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The impact of ecosystem services knowledge on decisions

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THE IMPACT OF ECOSYSTEM SERVICES KNOWLEDGE ON DECISIONS

A Dissertation Presented

by

Stephen Mark Posner

to

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of

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ABSTRACT

The need to protect diverse biological resources from ongoing development pressures is one of today's most pressing environmental challenges. In response, "ecosystem services" has emerged as a conservation framework that links human economies and natural systems through the benefits that people receive from nature. In this dissertation, I investigate the science-policy interface of ecosystem services in order to understand the use of ecosystem service decision support tools and evaluate the pathways through which ecosystem services knowledge impacts decisions. In the first paper, I track an ecosystem service valuation project in California to evaluate how the project changes the social capacity to make conservation-oriented decisions and how decision-makers intend to use ecosystem services knowledge. In a second project, I analyze a global sample of cases and identify factors that can explain the impact of ecosystem services knowledge on decisions. I find that the perceived legitimacy of knowledge (whether it is unbiased and representative of many diverse viewpoints) is an important determinant of whether the knowledge impacts policy processes and decisions. For the third project, I focus on the global use of spatial ecosystem service models. I analyze country-level factors that are associated with use and the effect of practitioner trainings on the uptake of these decision support tools. Taken together, this research critically evaluates how ecosystem service interventions perform. The results can inform the design of boundary organizations that effectively link conservation science with policy action, and guide strategic efforts to protect, restore, and enhance ecosystem services.

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CHAPTER 1: INTRODUCTION AND LITERATURE REVIEW

1.1. Big Picture Summary

The worlds in which scientists conduct research and policymakers craft policy do not always coexist. Scientists and policymakers can exist in two separate communities, with limited interactions and infrequent exchanges of information that lead to uninformed or ineffectual policy decisions. Without an effective interface between science and policy, good science may not have an impact on real-world decisions. This is a problem because past decisions based on a partial scientific understanding of how ecosystems benefit people have undervalued ecosystems and contributed to environmental decline. We can improve the environmental and long-term economic outcomes of policy decisions by doing a better job incorporating scientific information about the state or trend of environmental conditions and how development activities alter the flow of ecosystem services (the benefits that nature provides to people and that support human well-being).

When scientists go about their research without understanding how the knowledge they are producing might be used, the research occurs in a vacuum with limited connection to real policy decisions. This is especially important to consider given the urgency of global environmental problems today. Human activity now impacts the entire planet in drastic ways, leading to the claim that we have entered a period of time dominated by human-caused influence on the environment called the Anthropocene. One negative impact resulting from this extensive human activity is the widespread loss of biodiversity and ecosystem services (Millennium Ecosystem Assessment, 2005). This

eroding of the ecological support system of the planet threatens the stability and sustainability of economic systems (Kates et al., 2001).

If scientists want to make it easier for certain kinds of information to be incorporated into decisions, then they need a firm understanding of what makes knowledge useful to decision-makers. There is anecdotal evidence for how ecosystem services information has an impact on decisions, but there has not yet been much research about how or why this occurs (Laurans & Mermet, 2014; Laurans et al., 2013; Liu et al., 2010; McKenzie et al., 2014). Understanding what makes for effective interactions between scientists and policymakers will benefit those who produce ecosystem service knowledge as well as those who use it in decision-making.

A more effective science policy interface is needed for the protection, conservation, and restoration of ecosystem services (Nesshover et al., 2013). As coupled social-ecological systems undergo changes, we need an adaptive balance between real human development needs and the ongoing maintenance and health of Earth's life support systems (Weichselgartner & Kasperson, 2010). This requires policy informed by a sound scientific basis. At the same time, it calls for science that is use-inspired, policy-relevant, and defined according to the problems at hand.

The theory and practice of ecosystem service analysis provides a fertile landscape for work at the science-policy interface. Much work has been done on understanding the science of ecosystem services and how ecosystem structure can lead to ecosystem function with measurable benefits to people (Kremen, 2005). Other areas of research have focused on policy mechanisms, and how to mainstream ecosystem service concepts

into policy discussions (Daily & Matson, 2008; Fisher et al., 2008; Fisher, Turner, & Morling, 2009).

This doctoral dissertation investigates how ecosystem services science is incorporated into decision-making and what makes the production, distribution, and utilization of ecosystem services knowledge effective. My research uncovers how the use of ecosystem service science enhances the ability of decision-makers to understand 1) the benefits that people receive from nature and 2) how those benefits are likely to be affected by alternative decisions. Research objectives include to evaluate how ecosystem service interventions affect decision-makers and to identify enabling conditions for effectively translating ecosystem services knowledge into action. The results can inform how to be more effective in strategic efforts to protect and enhance ecosystem services, and guide the design of knowledge systems that effectively link conservation science to policy action.

The following overarching questions drive this dissertation research:

- 1) How do decision-makers use ecosystem service knowledge?
- 2) What is the impact of ecosystem service knowledge on decisions?
- 3) Why does ecosystem service knowledge have an impact? What factors (i.e. attributes of the science, the decision context, and the policy process) can explain the impact of knowledge?

1.2. Natural Capital and Ecosystem Services

This section describes the emergence of ecosystem services as a framework for the conservation of nature and biodiversity. I describe some of the ways this concept has

been put to use in various contexts, and discuss theoretical critiques and debates about the risks of embracing such an anthropocentric and economically based framework.

1.2.1. From Debut to Widespread Popularity

The concepts that underpin “natural capital” and “ecosystems services” emerged as early as the 1920s (Costanza & Kubiszewski, 2011). E.F. Schumacher in his popular book *Small is Beautiful* (1973) framed many of the general ideas and invoked the term “natural capital.” In the peer-reviewed scientific literature, Ehrlich & Mooney (1983) are credited with the first use of the term “ecosystem services”, while a paper by Costanza & Daly (1992) is one of the first to explicitly mention of the term “natural capital.” Subsequent contributions had significant influence on mainstreaming ecosystem service concepts, in particular: a research article by Costanza et al. (1997) that estimated the value of the world’s ecosystem services; work by Lovins, Lovins, & Hawken (1999) that described the implications of natural capitalism for economies, industries, and businesses; and books by Daily (1997) and Daily & Ellison (2002) that spoke of the growing recognition of ecosystems as capital assets, with diverse examples of how people have benefited from a natural capital approach to management and conservation.

More recently, the notion of ecosystem services has gained wider use among scientific, business, and political communities. The Millennium Ecosystem Assessment (2005) defined clear as well as less tangible, though still important, ecosystem service categories in terms of the benefits that humans obtain from nature. This global assessment describes four main categories of services. Provisioning services support the ways that people directly extract benefits from nature (i.e. food, timber, clean water). Regulating services moderate natural phenomena to people’s benefit (i.e. carbon

sequestration, flood control, decomposition). Supporting services provide a more fundamental basis for the Earth sustaining living things (i.e. photosynthesis and primary production, nutrient cycling). Non-material cultural services contribute to the cultural and social development of people (i.e. recreation, spiritual inspiration). Several other ecosystem service frameworks exist, for example The Economics of Ecosystems and Biodiversity study (TEEB), which modifies the MEA framework to more explicitly account for the role of biodiversity (Bishop, 2012; Kumar et al., 2010).

Today, the concepts of natural capital and ecosystem services have moved beyond scientific research. In the fields of law and policy, (Ruhl, Kraft, & Lant, 2007) describe detailed case studies that highlight how land use is a driving factor in the creation of ecosystem services, and how trade-offs between ecosystem goods, ecosystem services, and other desirable aims require that society choose what it wants from a set of possible future scenarios. In the business world, organizations such as Businesses for Social Responsibility (BSR), the World Business Council for Sustainable Development (WBCSD), and the World Resources Institute (WRI) have all developed approaches to measuring and assessing the ways that companies interact with ecosystem services, often through valuing the services in monetary terms.

Governments have also demonstrated interest in the ecosystem services concept, and in some cases have undergone national-level studies such as the UK National Ecosystem Assessment (Bateman et al., 2014; Watson, 2012). Other federal governments have integrated natural capital ideas and approaches into agency frameworks, as in the USDA Office of Environmental Services, the USGS public domain tool Social Values for Ecosystem Services (SolVES), and the EPA's unit on Ecosystem Services Research.

Meanwhile, The Economics of Ecosystems and Biodiversity (TEEB) and the Intergovernmental Platform on Biodiversity and Ecosystem Services (IPBES) indicate that natural capital continues to gain traction in international planning and governance settings as well.

1.2.2. Conditions for Ecosystem Services Science Success

A report on The State of Ecosystem Services by Cox and Searle (2009) found that fragmented knowledge on ecosystem services indicates a need for replicable and standardized projects. The authors of the report suggest the following four conditions for ecosystem service conservation success:

- clear science (about ecosystem services, interactions between services, and how proposed actions may affect services),
- defined benefits to people (with quantifiable values, and providing links from ecosystems to human well-being),
- a confined system (with clearly identified stewards, perpetrators of negative impacts, and service beneficiaries), and
- good governance (in terms of clearly defined ownership or tenure, a legal system, enforcement capacity, monitoring of impacts, and a functioning infrastructure to support and enable projects).

Several efforts have evolved in the process of establishing standardized scientific approaches to modeling and measuring ecosystem services, from early efforts like the Global Unified Meta-model of the Biosphere (Boumans et al., 2002) to more recent projects that incorporate machine learning and spatial ecosystem service flows such as ARtificial Intelligence for Ecosystem Services (ARIES) (Bagstad et al., 2013; Villa et al.,

2014). The Natural Capital Project with its Integrated Valuation of Environmental Services and Trade-offs (InVEST) family of tools, has demonstrated success in terms of replicable and standardized projects across a variety of political and cultural contexts (Tallis & Kareiva, 2009). Their framework for integrating natural capital into decision-making shows how biophysical models of ecosystems and the services they produce are combined with economic and cultural models to describe ecosystem service values. This information is then mediated through institutions, and decisions are made with impacts on ecosystems and ecosystem services (Figure 1.1).

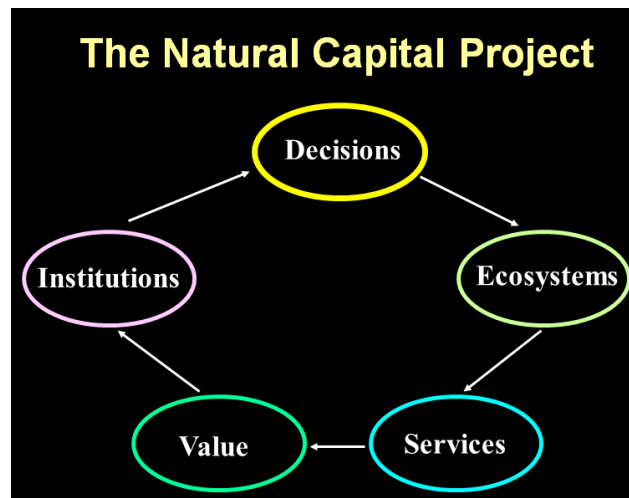


Figure 1.1: A Natural Capital Project framework for how ecosystem services can be integrated into decision-making. Based on Daily et al. (2009).

Calls have been issued for ecosystem service research that generates new policies, incentives, and institutions on a large scale (Daily & Matson, 2008; Daily et al., 2009). Kinzig et al. (2011) and Wunder, Engel, & Pagiola (2008) describe factors that influence the effectiveness of various policy approaches, including the cost of alternative built infrastructure and enforceable actions to ensure service provision. McKenzie and Irwin (2011) present a table of “enabling conditions for policy instruments” regarding

ecosystem service and biophysical attributes, producers of ecosystem services, beneficiaries, and costs and benefits. Wunder et al. (2008) compare payment for environmental services (PES) programs and find that more careful analysis is needed to understand how such policy approaches work. This dissertation builds on and extends this work by focusing on the enabling conditions for ecosystem service science to have influence.

1.2.3. Is Ecosystem Services More/Better/Different Biodiversity Conservation?

But does the concept of ecosystem services add anything more, better, or different to biodiversity conservation efforts? McCauley (2006) argues that ecosystem services is a new means to the same end goal of biodiversity conservation. He points out that ecosystem services can at times be “useful bargaining chips in specific conservation plans,” but he urges a return to the primary mission of protecting nature for its own sake and not necessarily to guarantee a profit. According to this view, ecosystem services may not provide anything other than a distraction from the previously established end goal of conserving biodiversity. And while an approach based on ecosystem services can be a means to achieving biodiversity conservation, there are situations in which it will not, such as increasing food provisioning services by replacing natural vegetation with crops, or maintaining a supply of freshwater by building a dam that impacts fish and countless other species (Reyers et al., 2012).

The viewpoint that ecosystem services is just a new means to the same end often compares the additional influence that can be achieved by an ecosystem service approach with the potential risks and drawbacks. Do ecosystem services detract from the moral arguments underpinning biodiversity conservation? Do they risk reducing people's

relationships with nature to purely economic terms? Supporters of this viewpoint raise these important questions and voice skepticism that ecosystem services can continue to live up to the expectations posed by significant recent interest and support from governments, conservation organizations, and academics.

Others feel that maintaining ecosystem services is a valid end in itself. Costanza et al. (1997) present a perspective based strongly on the protection of necessary ecosystem services as a worthy end goal. Maintaining ecosystem services, they argue, is the goal because these services are the life-support system for humans and the foundations of our economies. A focus on ecosystem services could ensure the sustainability of human welfare on the planet. The authors of this study consider “natural capital as essential to human welfare. Zero natural capital implies zero human welfare because it is not feasible to substitute, in total, purely ‘non-natural’ capital for natural capital.” Thus, the effort to conserve and protect ecosystem services needs to be a focused endeavor in and of itself.

Balmford et al. (2002) also frame the maintenance of ecosystem services as an end goal. They focus on the economic benefits of conservation efforts and how in most cases, when considering a full range of factors, the benefits of conservation outweigh the benefits of alternative decisions such as forest conversion. By maintaining ecosystem service flows first and foremost, we can ensure the sustainability of human communities that depend on these services. These arguments are based on connections between ecosystem services and human well-being, adopting an anthropocentric view of the importance of nature’s benefits to humans.

Lastly, there is the view that ecosystem services and biodiversity conservation are parallel goals to be done simultaneously but independently. A paper by Chan et al. (2006) supports this view and explores how well biodiversity and ecosystem services align within a conservation planning framework. They find that it is important to pursue the two goals at the same time. Targeting only biodiversity or ecosystem service goals can lead to trade-offs in which pursuit of one objective occurs at the expense of the other.

Broadening conservation goals to include both biodiversity and ecosystem services can “sustain critical services, open new revenue streams, and make conservation broad based and commonplace” (Chan et al., 2006). Indeed, many others, including Goldman & Tallis (2009) think that parallel goals have the potential to attract additional funding, engage a broader set of stakeholders, and establish conservation as more mainstream practice. Reyers et al. (2012) also conclude that the conservation community needs to move beyond the either/or debate to embrace both intrinsic biodiversity values and instrumental ecosystem service values in dealing with the problem of biodiversity loss.

1.2.4. Limitations to a Natural Capital Approach

Many within the conservation community and beyond voice concerns about the rapid rise in popularity of the ecosystem services concept. Critics point out potential conflict in situations that have unavoidable trade-offs between biodiversity and ecosystem services. Reyers et al. (2012) discuss the potential for biodiversity conservation to decrease the provision of ecosystem services, as well as how management designed to increase ecosystem services could have negative impacts on biodiversity. Others point out how emphasizing ecosystem services can have limited or

no effect on actual land management decisions if technological or other economic conditions are stronger drivers of behavior (Ghazoul, 2007).

Beyond the ecological and human behavior outcomes of ecosystem service applications, there are serious concerns rooted in ethical considerations, including changes in motivations for conserving nature (or not), the effect of commodification, and equity implications (Luck et al., 2012; Muradian et al., 2013). For example, Redford & Adams (2009) describe some of the potential problems with payment for ecosystem services schemes that arise when economic pricing of services from a landscape overshadow the intrinsic value that could motivate conservation. Market-based mechanisms can be a risky diversion from what some believe to be a more important ethical motivation to conserve nature (McCauley, 2006).

Commodification presents other issues as well. Ecosystem services based on social values can be difficult to price, such as the spiritual benefits of a forest. Commodification involves narrowing the complexity of ecosystems, and this can deny multiplicity in values (Kosoy and Corbera, 2010). Ecosystem services applications do not always acknowledge the multiple, potentially incommensurable values for nature that different stakeholder groups hold. Trade-offs in implementation across diverse socio-ecological contexts may require more deliberative methods for uncovering multiple value systems (Kosoy and Corbera, 2010). When the process of valuation compresses information about ecosystem attributes into a single monetary metric, it creates an unavoidable loss of information about ecosystems (Vatn and Bromley, 1994). This loss of information associated with pricing and valuation is nonrandom, thus valuation does not

reliably generate conclusive results about the “best” individual or collective welfare-enhancing decisions with regards to nature.

Equity considerations further limit the appropriate use of an ecosystem services approach. Ecosystem services can propagate or reinforce asymmetries in the distribution of costs and benefits among different groups. The distribution of some essential ecosystem goods and services (i.e. food) is unjust because overreliance on competitive market-based forces can lead to speculation, price instability, and inefficient allocation of essential resources (Farley et al., 2014). In protected areas and rural communities in particular, concentrated decision-making power leads to ecosystem service initiatives based on the needs and desires of some groups at the expense of others (for example, service providers can directly receive compensation more than service users, who may be excluded from receiving benefits of development) (Corbera et al., 2007).

Those in positions of power often define the legitimacy of ecosystem services knowledge. Different knowledge and value systems present in many ecosystem service contexts require deliberation and collective action to emerge from reciprocal interactions over time (Kolinjivadi et al., 2015). But this time-intensive process does not always occur. The complexity of power relations and institutional settings for ecosystem services projects can lead to a form of knowledge imperialism (Hardy and Patterson, 2012). Ecosystem services can become a form of projecting power onto developing world (Muradian et al., 2010). Indigenous groups may have long-established ways to sustain the forests upon which they depend. But ecosystem services may risk disregarding such cultural practices, along with traditional ways of knowing and valuing nature (Ernstson and Sorlin, 2013). The effectiveness of an approach based on ecosystem services depends

in part on the political forces at play and the structure of institutions for mediating information about values, expert and traditional sources of knowledge, and the distribution of costs and benefits across different populations (Muradian et al., 2013).

It is important to recognize that ecosystem services is only one way to understand nature and the relationships that people have with the environment (Luck et al., 2012). Researchers must consider the unique context and place of an application before deciding whether an ecosystem services approach is warranted. The degree to which ecosystem services conservation becomes a distinct goal depends on these specific considerations and limitations, as well as the general ways science interacts with policy.

1.3. The Science-Policy Interface

This section presents a review of the “science-policy interface” literature. It describes theories of how policy is formulated, how knowledge is used by policymakers, and the importance of spanning the boundary between science and policy worlds. It ends with theories of how knowledge links to action that provide intellectual underpinnings for this dissertation.

The science-policy interface refers to the intersection of scientific assessment and policy decision-making (Watson, 2005). Van den Hove (2007) defines these interfaces as “social processes which encompass relations between scientists and other actors in the policy process, and which allow for exchanges, co-evolution, and joint construction of knowledge with the aim of enriching decision-making.” Research in this area is concerned with the role and use of scientific evidence in policy formulation, the ways

that scientific research efforts interact with policy processes, and the characteristics of science that can lead to effective policy.

The conceptual separation of science and policy reflects how, in a conventional sense, scientists and policy-makers have conducted their work in largely separate spheres. The interface exists as a conceptual boundary between two separate communities (Kerkhoff & Lebel, 2006). On one side of the science-policy interface are scientific communities that investigate, organize, and report knowledge. On the other side, there are policy communities that make far-reaching decisions based on knowledge, values and beliefs, and perceived priorities. Huitema & Turnhout (2009) describe how the cultures of science and policy are often perceived to be very different, from the ways in which they communicate, to the interests they pursue, to their fundamental levels of objectivity or subjectivity.

The boundary between science and policy can range from sharp and impermeable to more blurred and porous (Guston, 2001). The interface is often mediated by institutions called boundary organizations that determine processes of decision-making among groups of stakeholders with diverse values. One major disagreement within the literature has to do with whether it is better to attempt to blur the lines to overcome the differences between communities of scientists and policymakers (and if so, how best to do so), or rather to maintain firm boundaries between the worlds of science and policy so that each can function independently without undue interference by the other.

For example, Leiss (2000) argues strongly for separate and independent scientific assessments of policy options. He feels that scientific research agendas should not be constrained by political conflicts, and that a critical role for policy-makers is to manage

risk based on information provided by independent science. If a government becomes overly involved in conducting scientific research itself, then the public could perceive it as an “interested party” and potentially question the legitimacy of its scientific efforts.

Leiss (2000) rightly points out how problems of risk management have become increasingly important as human population growth and pressure on the environment have led to natural resource issues in which we no longer have the same margin of safety between sustainable and unsustainable resource use. Thus, the process of developing policy must include managing scientific information to compare relative risks of options. He stresses, though, that policy choices cannot be dictated by scientific study – that a firm boundary between science and policy can ensure risks are properly managed based on objective scientific truths. This would be consistent with the “pure scientist” role defined by Pielke (2007), in which distance between scientists and policy-makers helps maintain objectivity of the science.

Others disagree and argue for a more co-mingled process of joint production of research between scientists and policy-makers (Watson, 2005; Watson, 2012; Driscoll, Lambert, & Weathers, 2011; Cash et al., 2003; Lemos & Morehouse, 2005). In a co-production scenario, scientific experts and decision-makers come together to collaboratively produce research outputs that can be used in formulating policy. Some claim that it is virtually impossible for researchers to remain insulated from the world of politics when there is strong political debate about the nature of the problems being addressed (Hirsch & Luzadis, 2013; Huitema & Turnhout, 2009). For this reason, an “honest broker” role for scientists may be most appropriate, in which researchers expand the range of policy options by communicating with policy-makers and providing

scientific knowledge about options and their implications (Pielke, 2007). If scientists were to reduce the range of policy options and align with particular policies, they would be characterized more as “issue advocates” (such as the potential shift within the IPBES from more of an investigative function to an advocacy one described by Perrings et al., 2011). Scientists could also play an “arbiter” role if they engage with policymakers to provide expert scientific opinion but steer clear of the responsibility to make decisions about which policies to adopt (Pielke, 2007).

Even while disagreement exists about the extent to which the worlds of science and policy interact, there is general agreement about important factors that constitute effective science with regards to its interaction with policy. An open and transparent peer-review process for science is understood to be an important element for maintaining trust of scientific knowledge (Watson, 2005; Watson, 2012; Cash et al., 2003; Reid et al., 2009). Jones, Fischhoff, & Lach (1999) represent another widely-held view in the science-policy literature: that research results should be relevant to current issues of interest if they are to be useful.

Ultimately, it is unrealistic to expect an impermeable boundary be maintained between science and policy. At the same time, it is threatening to the integrity and credibility of science for scientists to be directly involved in advocating for political issues. In striking a balance, scientists must strive for an “honest broker” role in which they acknowledge and interact with the world of policy, but conduct research that expands the scope of choices to policymakers (rather than narrows it), and systematically evaluates the implications of the choices. In this way, the science-policy interface can be

characterized by a flow of information more about policy options but less about recommendations, with limited value-based judgments attached to any particular option.

The following topics from the academic literature represent different theoretical lenses through which researchers conceive of the science-policy interface and the processes through which knowledge and policy action are connected. The discussion provides general background and informs the theory of change and methods used in this doctoral research.

1.3.1. Advocacy Coalition

Sabatier & Jenkins-Smith (1993) have advanced an advocacy coalition framework as one approach to understanding how science is translated into policy. This framework acknowledges the role of active attempts to influence government policy decisions (advocacy) and focuses on the interactions of actors and institutions (coalitions) in this process (Weible et al., 2011). Sabatier and Jenkins-Smith point out how policies and programs can be understood as “sets of value priorities and causal assumptions about how to realize them” (Sabatier & Jenkins-Smith, 1993). Thus learning becomes a centrally important process as a mechanism for altering a belief system.

Enabling conditions for policy-oriented learning may include: an intermediate level of conflict between competing belief systems (too little conflict does not provide the tension needed to catalyze action while too much leads to irreconcilably incompatible worldviews); analytical tractability (accepted quantitative data and theory about natural systems that can convince the need for policy); and a forum that is professional and conducive to different authoritative figures vetting their perspectives.

1.3.2. Complexity

Elliott & Kiel (1999) and Geyer & Rihani (2010) present a foundation for social science and public policy based on complexity. They describe complexity in relation to modern and post-modern views on science and policy. The modernist approach suggests that knowledge is essentially order. A social system that increases knowledge also increases order and thus predictability and potential for control and management. The world is objectively knowable and more knowledge should be a policy goal. On the other hand, in the post-modern view, the world is relational and there is no way to accurately know the “right” policy because all policies can be understood from different, sometimes mutually incompatible perspectives – policy is always contested and is a response to an unpredictable world.

The complexity worldview appreciates that science can be used to know more, but there are always limits to knowledge, and the world is dynamic and full of uncertainty. Rather than achieving some end goal, policy needs to steadily learn to adapt to changing conditions. More knowledge is useful, but not necessarily effective due to the emergence of new issues from the complex interactions of many, dynamic parts of any system. According to this framework, flexibility and adaptive management in policy processes are key attributes to pursue. Berkes (2009) further suggest that adaptive resource management and adaptive governance, in which science and policy are co-mingled, are critical to the resilience of social-ecological systems.

1.3.3. Knowledge Transfer

A static model of how science translates into policy involves the transmission of scientific facts and results from scientists to policymakers. Policymakers then use the scientific information they receive to make decisions and formulate policy (Reynolds, 2007). Knowledge transfer suggests that knowledge exists as a tidy bundle that can be delivered to the appropriate user in a well-defined time frame and with a clear sequence of discrete steps toward a policy outcome. In reality, a strictly linear conceptual model of science transferring knowledge to policy does not fit well. Rather, the boundary between science and policy may be fuzzy, with knowledge use as more of a dynamic process than an event that takes place at a moment in time (Miller, Jasanoff, & Long, 1997). A science-policy interface can involve interactions that are iterative, more long-term, and evolve over time as relationships between scientist, policy-maker, and citizens, giving rise to the idea that organizations can be boundary-spanning between the worlds of science and policy (Reynolds, 2007).

1.3.4. Boundary Organizations

Boundary organizations serve to bridge the science and policy worlds. White, Corley, & White (2013) describe three main aspects of boundary organizations. First, they facilitate the creation of boundary objects that serve as mutually acknowledged and used artifacts between science and nonscience communities (i.e. computer models, maps, patents). Second, they include scientist and decision-maker participants mediated by professionals. Third, they are accountable in different ways to both the political and scientific systems.

Boundary organizations can facilitate the joint production of ecosystem service knowledge, which involves knowledge generators and users working together to answer management and policy questions. When decision-makers are integrated as part of the scientific enterprise, they can go beyond simply interacting with scientific knowledge – they can become connected to the scientific process as co-investigators. As a result, they may come to develop deeper levels of trust of the knowledge outputs and the underlying science upon which it is built.

Scientific research processes that are in collaboration with decision-makers have high potential for influencing decisions (Rowe & Lee, 2012). Collaborative processes that span the boundary provide a forum for scientists and policy-makers to define and answer questions together. Karl, Susskind, & Wallace (2007) and Andrews (2002) stress “joint fact finding” and two-way dialogues as key processes for an effective interface of science and policy. Continuous, iterative processes that involve both scientists and policymakers can create mutual understanding of problems and increased likelihood of ecosystem service information being used in crafting solutions (Reid et al., 2009; Nesshover et al., 2013; Ruckelshaus et al., 2013).

Numerous researchers have found trust to be an important part of effective participatory science-to-policy processes (Dalton, 2005; Reed, 2008; Voinov & Gaddis, 2008). Strong social capital can facilitate cooperation among scientists and policymakers by enhancing: trust relationships; reciprocity and exchange; common rules and norms; and connectedness of individuals in networks and groups (Pretty & Smith, 2004; Pretty & Ward, 2001). Boundary organizations can cultivate trust that helps actors relate quickly

and easily, increasing the chances that any improvements or successes achieved last beyond the timeframe of the project (Reid et al., 2009).

1.3.5. Knowledge Utilization

The knowledge utilization and agenda setting literature focus explicitly on the role of scientific knowledge in policy development. Hinkel (2011) considers the utilization of knowledge in indicators that can be used to monitor policy performance over time or to justify action/inaction. There is general agreement in the literature that policymakers can utilize scientific knowledge in distinct ways. However, the terms used to describe different types of knowledge utilization may differ. Weible, Pattison, & Sabatier (2010) describe learning, political, and instrumental functions of scientific information in policy, while Laurans et al. (2013) describe similar functions in terms of informative, technical, and decisive uses. Of the different uses of scientific environmental knowledge described in the literature, there is overlap and general agreement about three clear ways that policymakers utilize knowledge: to build support for their agendas, learn new information about policy problems and the range of solutions, and/or make specific decisions based on scientific information. McKenzie et al. (2014) expound these three primary ways that decision-makers utilize knowledge, based on an extensive review of the science policy interface literature (Amara, Ouimet, & Landry, 2004; Haas, 1992; Kingdon, 2011; Lavis et al., 2003; Lee, 1993; Lester, 1993; Oh, 1996; Pannell, 2004; Roux et al., 2006; Tomich et al., 2004; van Kerkhoff, 2006):

- 1) Instrumental / problem-solving use – where decision-making or problem-solving directly uses knowledge from findings and recommendations.

2) Conceptual / indirect use – where ideas and concepts filter into thinking, debate and dialogue; knowledge provides new ideas, theories, interpretations or hypotheses about the issues and facts related to decision-making context, eventually catalyzing change in actions.

3) Symbolic / strategic use – where practitioners and decision-makers use knowledge to legitimate their views.

Groot et al., (2010) find that an ecosystem service approach has changed the terms of discussion and been most effective in the ‘conceptual enlightenment’ function of influencing decisions. Hirsch (2011) further indicates the importance of indirect use of knowledge, highlighting how trade-off thinking shapes language in policy dialogues and ideas about complex conservation issues. While all three uses can be present in a given policy or issue context, decisions-makers may utilize ecosystem services knowledge most effectively through the framing of discussions and debates. Weiss (1977) and Mitchell et al., (2004) claim that such indirect influence of information on policy development occurs by affecting who participates in discussions, how discussions are framed, and what issues receive attention and visibility in discussions.

The science policy interface literature on knowledge utilization also sheds light on key factors that influence the uptake of science. Landry, Amara, & Lamari (2001) examine the case of social science research in Canada and find that the behavior of scientists and decision-making contexts have more influence than the attributes of the knowledge itself. This indicates the importance of focusing on knowledge-to-action processes and relationships rather than simply outcomes of either scientific or policymaking efforts. Andrews (2002) and Karl et al. (2007) highlight how ecosystem

service knowledge that is jointly produced by scientists and stakeholders is more likely to influence the way people conceptually understand the issues.

Anderson (1975) provides a similar theoretical lens for understanding policy processes. The policy stages framework they describe is helpful for disentangling the complex processes involved in policy and focusing on distinct subprocesses such as problem identification, agenda setting, adoption, implementation, and policy evaluation. This framework is applied to environmental policy by Tomich et al. (2004) in describing the following stages of an environmental issue cycle: perception by pioneers of problem; lobbying by action groups; increasing acceptance of impacts; debate cause and effect; inventory options; negotiate prevention or mitigation of impacts; implement, monitor, and enforce.

At which of these stages does ecosystem service science have the most influence? The ‘conceptual enlightenment’ model of knowledge utilization suggests that ecosystem service science would have most influence in the earlier stages. But recent approaches such as action research and innovative participatory processes draw into question the utility of this linear model of subprocesses (Kerkhoff & Lebel, 2006). The stages framework has serious limitations. For example, it is not a causal model in that it lacks identifiable linkages, drivers, and influences that exist within and between the distinct subprocesses. The policy stages framework is also inherently top-down and legal-focused, and so is likely to overlook important elements outside the purview of legislative processes – “behind-the-scenes” elements that in reality may be more important to how policy decisions are made.

1.4. Linking Knowledge to Action

This dissertation draws from these theoretical roots and builds on the ideas about knowledge systems described by Cash et al. (2003). In focusing broadly on how science and technology can be harnessed for sustainability, this research considers the networks and organizations of programs, institutional arrangements, and policies that comprise knowledge systems. Effective knowledge systems apply mechanisms that facilitate communication, translation, and mediation across boundaries between scientists and policymakers. Cash et al. (2003) identify three important features of the knowledge produced by scientific efforts:

- relevance (applicability of scientific assessment to the needs of decision-makers),
- credibility (“scientific adequacy of technical evidence and arguments”), and
- legitimacy (perception that the production of information and technology has been unbiased and respectful of diverse viewpoints).

Rowe & Lee (2012) describe a theory of change for how science has an impact on policy. They suggest that knowledge that is jointly produced by scientists and knowledge users is more likely to be relevant, credible, and legitimate, and thus more likely to have influence on real world decisions. These attributes can reinforce one another, but often they cannot all be optimized. Sarkki et al. (2013) further unpack the trade-offs between relevance, credibility, and legitimacy. For example, when a scientific output includes information about uncertainty, it can be viewed as more credible. But this may detract from the relevance to policymakers, who may find it difficult to understand or incorporate into policy. Similarly, a thorough and high quality scientific assessment takes

time, as does a legitimate process that takes into account plural perspectives. The time required to develop the credibility or legitimacy of scientific outputs creates a trade-off when policy work demands a rapid response.

Many opportunities exist for rich interaction between science and policy communities. This dissertation contributes to the growing body of literature that studies these rich interactions in detail in the context of ecosystem services and how knowledge links to action. In the following pages, I explore what constitutes an effective science-policy interface by: evaluating the impact that ecosystem services assessments have on decision-makers; analyzing global use of ecosystem service models; and testing how relevance, credibility, legitimacy, and other proposed conditions facilitate the impact of ecosystem service science.

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CHAPTER 2: EVALUATING THE IMPACT OF ECOSYSTEM SERVICE ASSESSMENTS ON DECISION-MAKERS

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2.1. Abstract

Ecosystem services are declining in many places and are impacted by land use decisions. Ecosystem service assessments can affect the capacity of decision-makers to make conservation-oriented land use decisions, but the actual impact of assessments is almost entirely unstudied. We conducted a difference-in-differences analysis of the impact of an ecosystem service assessment in California on decision-maker understanding of and attitudes about ecosystem services. We used surveys to compare “pre-” and “post-” differences in outcomes among two treatment groups in counties where assessment occurred and a comparison group without such assessments. Mixed methods included fitting regression models to estimate the treatment effect of the assessment and conducting interviews and direct observations to further understand how decision-makers changed their minds in response to the assessment. Regression results showed small increases relative to a comparison group in decision-maker understanding

of ecosystem services and perceived relevance of ecosystem services to their work. Analysis of the interviews confirmed that decision-makers became more aware of specific ways that ecosystem services could be used in conservation and development decisions, and believed that ecosystem services would improve the outcomes of land use decisions. Further impact evaluation studies of this type that estimate a counterfactual and explore rival explanations for observed outcomes are needed to build evidence for how ecosystem service projects impact relevant decision-makers and, ultimately, outcomes for environmental and human well-being.

2.2. Introduction

Land use and land management decisions have significant impacts on ecosystems and ecosystem services (ES), the valuable goods and services that ecosystems provide to people (Daily, 1997; Polasky et al., 2011). Increasingly, efforts to conserve, protect, or restore ES aim to influence land use decisions so that they incorporate information about the values of ES (Chan et al., 2006; Daily et al., 2011; Goldman & Tallis, 2009). Efforts to incorporate ES information into decisions rest on basic assumptions that this information will improve decisions and result in improved environmental and human well-being outcomes.

But there is a lack of sound evidence about the impact ES information has on decisions, or how decision-makers use ES knowledge (Laurans et al., 2013; Mermet et al., 2014). Many valuation studies mention prospective or intended roles for ES knowledge in terms of informative, technical, or decisive uses, but rarely do these studies describe actual use (Laurans et al., 2013). In a survey of researchers, Fisher et al. (2008)

found that ES research was used to inform policy agents, support policy initiatives, and directly influence government policy and investment. A recent review of three international case studies describe similar ways ES knowledge is used: *conceptually* to raise awareness and reframe dialogues, *strategically* to build support for plans or policies, and also *instrumentally* to make specific decisions (McKenzie et al., 2014). If conservation science is to inform improved land use decisions, it is critical to better understand what difference ES knowledge makes to land use decision-makers (McKenzie et al., 2011).

According to theories of the science-policy interface, knowledge has an important role in shaping decisions. Theory suggests that decision-makers are more likely to trust and use knowledge that they perceive as salient (i.e., relevant to the needs of decision-makers), credible (i.e., based on expert, reliable science), and legitimate (i.e., unbiased and inclusive of diverse perspectives) (Cash et al., 2003; Cook et al., 2013; Keller, 2010). A simple, linear model of policymaking described by Meier (1991) includes a role for knowledge early in the policy process, when it can affect the understanding and attitudes of policymakers. The more complex stages model (Grindle & Thomas, 1991), the policy streams model (Kingdon, 2011), and the advocacy coalition model of policy processes (Sabatier & Weible, 2014) all portray a similar key role for knowledge. These models share general components in common. Decision-makers and other stakeholders: perceive a problem, gather and evaluate knowledge about the problem and proposed solutions, and acknowledge the need to act on policy options.

Knowledge about the value of ES could thus be valuable as an early lens for identifying problems, a framework for crafting solutions and building support, and a tool

for evaluating proposed policy options. For ecosystem services specifically, a conceptual framework first presented by Ruckelshaus et al. (2013) and built upon by Posner et al. (in review) describes several pathways through which knowledge impacts policy decisions (Figure 1). Our study here mainly focuses on pathway 2, when ES knowledge helps shape the minds of decision makers by raising awareness and providing an ES focus for stakeholders. We also describe the emergence of pathway 3, through which decision makers and stakeholders build support for particular policy options and use language related to ES as a frame within policy dialogues. Lastly, we investigate the potential for pathway 4 and assess how decision makers envision using ES information to evaluate projects, compare options, and design new policies and plans.

The health, policy, and international development fields have long included systematic impact evaluation research, and now researchers and practitioners in conservation increasingly recognize the need for improved evidence of impact (Ferraro & Pattanayak, 2006; Fisher et al., 2013). The complexity and scale of real world social-environmental interactions has made rigorous and quantitative evaluation of impact in conservation difficult, but recent research is moving beyond anecdotal evidence and testing specific causal mechanisms through which impact may occur (Andam et al., 2010; Arriagada et al., 2012; Ferraro & Hanauer, 2014b; Miteva et al., 2012; Naidoo & Johnson, 2013; Pfaff et al., 2008). In order to understand how conservation programs and projects lead to improved outcomes for biodiversity and well-being, these studies use control groups and statistical matching to estimate impact (Ferraro, 2009; Margoluis et al., 2009).

Our study complements this growing body of work focused on the impact of conservation policy on environmental outcomes (pathway 5 in Figure 1). Our work here focuses on impact at a different, earlier part of the policymaking process – when ES knowledge has an impact on the minds of those proposing and making policy decisions (pathway 2 in Figure 1). We aim to detect whether information about the value of ES changes the capacity of natural resource managers and conservation decision-makers in California to make conservation-oriented decisions. In the process, we evaluate the importance and impact of ES knowledge as a resource for decision-makers.

Specifically, we ask: do ES valuation projects and the knowledge they generate impact local decision-makers' 1) understanding of ES and natural capital concepts, and 2) attitudes about conservation and planning approaches based on these concepts? We follow ES assessments in two counties in California, and use quantitative methods to compare changes in decision-maker understanding and attitudes with those in neighboring counties without assessments. We also use qualitative methods to explore why understanding and attitudes did or did not change. Tracking change in decision-makers and their capacity to consider ES is vital in order to link scientific knowledge with action, and to understand the difference that ES information may make.

2.3. Methods

“Healthy Lands, Healthy Economies” is a regional initiative designed to demonstrate the economic value of conservation in Sonoma, Santa Clara, and Santa Cruz Counties in California (herein referred to as the initiative). One of the goals of the initiative is to measure the tangible effect that protecting natural areas has on local and

regional economies. The multi-year project has resulted in ES valuation reports for Santa Clara County (Batker et al., 2014), Santa Cruz County (Schmidt et al., 2014), and Sonoma County (forthcoming) that frame the natural resources of each county as capital assets requiring investment in order to maintain a flow of economic benefits. The project team consists of ecological economists, ecologists, and conservation planners who work to identify and quantify the economic and community benefits achieved by investing in working lands, natural areas, and water resources in the greater San Francisco Bay Area.

We used a mixed methods approach to evaluate the impact of the initiative and the subsequent ES valuation reports on decision-makers in Santa Cruz and Santa Clara County (Bamberger, 2012; Creswell, 2009; Wholey et al., 2010). This approach allowed us to consider multiple sources of quantitative and qualitative evidence in our analysis and construct a more complete description of impact. Our overall approach involved before-after analysis of decision-maker understanding and attitudes in a treatment group in the two counties and a comparison group composed of decision-makers in surrounding counties that were not part of the initiative.

2.3.1. Quantitative Methods

We surveyed individuals in two treatment groups (Santa Clara and Santa Cruz counties, where the ES valuation study was conducted and reported) and a comparison group (8 neighboring counties, where no county-wide ES valuation studies were occurring). We administered the survey electronically using Survey Monkey before the initiative was launched in Winter 2013 and after release of the final county-level ES valuation reports in Winter 2015. Survey respondents were land use and conservation

decision-makers identified by our team as the intended audience for the initiative (for example, general managers of water districts, county planners, and executive directors of conservation NGOs).

In the comparison group of 63 individuals, we received 10 responses to both the pre- and post- survey (16% response rate). These 10 individuals were from Alameda, Contra Costa, Marin, Monterey, Napa, San Mateo, San Luis Obispo, and Solano Counties. In each of the Santa Cruz and Santa Clara groups, we received 9 responses to the pre- and post- surveys (18% response rate). Response rates were higher for the pre-survey, but fewer of these individuals responded to the post- survey and some initial respondents moved jobs or locations during the initiative. We only used survey results for individuals who responded to both the pre- and post- survey to maintain a consistent sample. Our data are based on 28 pre- and 28 post- survey responses: a total of 20 in the comparison group, 18 in Santa Cruz, and 18 in Santa Clara. The first round of surveys informed the design of interview questions used in the qualitative analysis.

The survey consisted of open-response and 5-point Likert-scale questions that we used in the following quantitative analysis (Appendix 3). For outcome variables, we used mean responses to seven survey questions with standard errors (all part of the first compound survey question): 1) relevance of ES to one's organization and 2) relevance to one's work; 3) credibility and 4) legitimacy of ES knowledge produced by the initiative; 5) understanding of ES; and 6) the capacity within one's county to monitor impacts to ES, and 7) implement policies or plans about ES.

We used a difference-in-differences approach to estimate the effect of the initiative on decision-makers (Gertler et al., 2010; Khandker et al., 2010). We calculated

the before-after differences for outcome variables in the treatment groups; calculated the before-after differences for the same variables in the comparison group; and estimated the average treatment effect as the difference between before-after changes between each treatment group and the comparison group (Table 1).

We then specified time-series linear regression models:

$$Y_{it} = \beta_0 + \beta_1 P_{it} + \beta_2 G_{it} + \beta_3 (P_{it} \times G_{it}) + \varepsilon_{it}$$

where Y_{it} is a decision-maker outcome of interest, P_{it} is a dummy variable for time period where pre-initiative periods are 0 and post-initiative are 1, and G_{it} is a dummy variable for group where comparison group is 0 and the treatment county is 1 (Branas et al., 2011). We defined the term $P_{it} \times G_{it}$ as the interaction between a pre-initiative and post-initiative difference for each decision-maker. The coefficient for the interaction term β_3 is identical to the difference-in-differences term calculated above, and estimates the average effect of the initiative on the outcome (Meyer, 1995). All data analysis was performed in R (R, 2011).

2.3.2. Qualitative Methods

We used a case study approach to gather, organize, and analyze data (Yin, 2009). We conducted direct observations of 10 initiative workshops and meetings in Fall 2012 through Winter 2013. This data collection technique allowed us to systematically observe decision processes and early dialogue using a structured, pre-designed observation record form (Appendix 1) (Taylor-Powell & Steele, 1996). We observed decision-makers and scientists in their natural settings as they engaged with ES and natural capital concepts during the process of the ES valuation study.

We conducted 11 semi-structured interviews with stakeholders before the initiative was launched and 12 interviews after the county reports were published (23 total interviews) (Appendix 2). Interview questions built upon the results of the pre-initiative survey. We used content analysis on the interviews and direct observation records to understand the impact of the initiative.

2.4. Results

2.4.1. Quantitative Results

Difference-in-differences estimates on the outcome variables showed mixed results (Figure 2, Table 2). Most estimates of impact from the regression models were weak. We emphasize two results with a p-value ≤ 0.15 (because our small sample sizes make it difficult to detect differences). In Santa Clara County, we found an increase in the perceived relevance of ES to one's work. In Santa Cruz, our results show an increase in the understanding of ES. Other relationships were non-significant, though, including positive estimates of impact in Santa Clara for the perceived relevance and legitimacy of ES and the capacity to monitor ES; and in Santa Cruz for perceived credibility of ES and relevance of ES to a person's work, understanding of ES, and capacity to monitor ES. In both counties, non-significant negative impacts were found for the capacity to implement policies or plans about ES.

Looking more closely at the raw data, we examined the outcome variables with $p \leq 0.15$ from our regression results. For the survey questions we used to measure outcomes, we found changes between the pre and post periods in the percentage of respondents who agreed or strongly agreed (Table 3). For example, this percentage for

understanding of ES had the largest increase of all outcomes in both counties relative to comparison group: in Santa Clara, from 55.6% pre-initiative to 77.8% post-initiative; in Santa Cruz, from 33.3% to 77.8%; and in the comparison group, no increase from 90%. Also, the percentage of respondents who agreed or strongly agreed that ES is relevant to their work went up from 77.8% to 88.9% in both Santa Clara and Santa Cruz, but went down from 100% to 90% in the comparison group. The decrease in perceived capacity to implement policies or plans about ES is also evident in the raw data. In Santa Clara, the percentage of respondents who agreed or strongly agreed declined from 77.8% pre-initiative to 55.6% post-initiative; in Santa Cruz, from 55.6% to 33.3%; in the comparison group, from 50.0% to 40.0%.

2.4.1. Qualitative Results

The results of our qualitative analysis provide additional insight into the impact of the initiative. Again, there were some mixed responses among decision-makers, but the interviews uncovered stronger evidence of impact. The discussion that follows considers why we observed different impacts using quantitative and qualitative approaches.

Decision-makers understood ES better, but found it a new concept for themselves and the public

The interviewees used more technical language about ES and demonstrated increased understanding of ES topics post-initiative. Interviewees reported being more comfortable discussing ES concepts. Several people described ES as “a new way of thinking” and “a new concept for people,” even “a new paradigm.” In pre-initiative

interviews, they expected the initiative to “bring added information” to land use decisions. Post-initiative, interviewees confirmed that this was occurring. Decision-makers felt that the new information provided in the initiative could inspire a different way of doing conservation with better environmental and long-term economic outcomes.

Decision-makers were initially skeptical of ES, but acquired ways to deal with their uncertainties

In 3 of the 12 follow up interviews, decision-makers reported initial skepticism about ES, but over the course of the initiative came to perceive ES as valuable for informing decisions. Also, 4 interviewees pre-initiative described the potential for information about ES to be either good or bad, depending on how the information would be used. Post-initiative, there was less concern about the potential mis-use of ES information, for example, to inflate property values prior to open space acquisition.

Decision-makers had questions about the often-wide ranges of ES value estimates. For example: “to what extent can the ES valuation numbers be accepted as empirical and exact figures, versus just starting points for needed conversations?” Half of the interviewees post-initiative reported not having any problem with the value ranges. Others still felt the ranges could hurt the credibility of the ES value estimates, but most accepted the value ranges as useful for long-range regional-scale planning.

Several decision-makers mentioned that their organizations were considering ways to internally evaluate the ES value estimates (for example, by having staff ecologists review how the ES valuation reports determined which ES were provided by different land use types). This indicated engagement with the reports. Vetting of the ES

assessments, either internally within an organization or publicly, could lead to more co-production of useful ES knowledge (Cutts et al., 2011; Roux et al., 2006).

Decision-makers used or intended to use the ES knowledge conceptually, strategically, and instrumentally

Of all the interviewees, only 3 of the total 23 were skeptical about ES and their utility in land use decisions. There was widespread agreement about the value of ES knowledge for framing conversations about nature and highlighting trends in environmental quality. One interviewee described how the initiative “painted a picture to demonstrate how the value of each [ES] affects people in real ways.” A representative from a funding organization described how the initiative was helping ES become a regular part of their “lexicon.” These results indicate conceptual use of the ES knowledge.

There was also clear evidence of strategic knowledge use. Decision-makers felt that the ES knowledge provided by the initiative would be used: to inform conversations about the potential formation of a new open space district; to build public support for conservation funding measures; and to “change legislators’ thinking about the value of state parks or open space.” There was also mention of building support within the business community because of how nature “provides nice views and recreation, attracting and retaining a qualified workforce.”

Pre-initiative interviews found that people held a range of general ideas for instrumental use of ES knowledge. Post-initiative, people could bring these ideas into sharper focus. Decision-makers described more specific intended uses: to inform coastal

management and identify vulnerable areas for planned retreat vs. other actions; to “evaluate priorities and options for watershed stewardship projects (such as habitat improvement, invasive species removal, fish barrier removal, etc.)”; and to help decide whether to build desalinization plants to maintain water supply. In particular, people felt the dollar value estimates from the initiative would improve cost/benefit analysis in infrastructure decisions. In pre-initiative interviews, people were interested in a more integrated, coordinated approach to decisions about conservation and infrastructure (which are often thought about separately in terms of funding and jurisdiction). Post-initiative interviews found this interest was bolstered by the ES valuation reports. There was clear intention to use ES knowledge to develop a more balanced investment between “green” and “grey” infrastructure.

Decision-makers believed the initiative would help shift patterns in land use and development

An important component of our evaluation study was to ask decision-makers to consider what would occur in the absence of any ES value assessment in their counties. Interviewees all generally agreed that without the initiative, things would continue in a “business as usual” way, meaning that “there would be a continued undervaluing of green infrastructure and an overemphasis on grey, built infrastructure.” In pondering this hypothetical question, all interviewees felt that without the initiative, there would be ongoing loss of nature in the region. Many also mentioned opinions that the initiative would facilitate including ES in cost-benefit analyses for projects and shift peoples’ perspectives on conservation.

2.5. Discussion

We found strong evidence of the initiative's impact with our interviews, but weaker evidence of impact with our difference-in-differences analysis. By triangulating among qualitative and quantitative data, we found slightly different results in our mixed methods impact evaluation. Overall, the strongest impacts post-initiative were in how decision-makers came to understand ES more and envisioned more specific ways in which they could use ES knowledge in their work.

A few factors could explain the different results between Santa Clara and Santa Cruz Count. There are differences in natural resource management between counties that affect the dissemination and potential use of ES information (i.e. Santa Clara County has one large consolidated water district, while Santa Cruz County has many small individual water districts). Also, there are small differences in how and when the initiative occurred in either county. The Santa Clara report on Nature's Value was completed three months before the Santa Cruz report. And while a similar process was conducted in both counties, there were inevitable differences in two separate valuation processes. Lastly, ES knowledge can be expected to have different impacts within the varied cultures of the Santa Clara and Santa Cruz counties (one county is a technology business hub while the other is a coastal recreation destination).

The quantitative and qualitative results are complementary, but the differences need to be examined in the context of the entire study. The quantitative results from our survey indicate the initiative coincided with an increase in some decision-makers' general understanding of ES and perceived relevance of ES to one's work. In considering rival

explanations for the observed outcomes, we highlight four reasons why the difference-in-differences results were not very significant and posit how likely each is.

First, the quantitative results could mean that the initiative did not make a difference. This is an unlikely reason for the non-significant results. The qualitative results indicate that the initiative did make a difference, as they confirm a growing understanding of ES and show that decision-makers acquired insights into specific ways they could use ES knowledge to inform decisions. However, the qualitative results show that the initiative's impact was stronger for certain outcomes, such as understanding of ES, than for others, such as credibility of ES valuation methods. This heterogeneous impact across the outcomes we measured would produce some non-significant results in our quantitative study, but the qualitative methods uncovered clear impacts from the initiative.

Second, concurrent factors could have influenced the observed outcomes in decision-makers. ES has been an increasingly popular topic in conservation, and other state or national ES programs could have influenced the comparison or treatment groups (and we assume it would influence them all equally). Another factor was the record-setting drought that consumed the attention of many California decision-makers during the timeframe of the study. While some aspects of ES were relevant to freshwater provision, ES as a whole was not as primary an issue as water shortage and management issues. Lastly, a spillover effect from treatment counties to nearby comparison counties could also have influenced our survey results. There were undoubtedly pre-existing relationships among people in the northern California conservation community. Since

this spillover would equalize outcomes between groups, it is a likely explanation of some of the non-significant difference-in-differences results.

Third, we acknowledge low sample size, selection bias, and other statistical limitations. By the time we collected post-initiative data, we had small sample sizes of individuals who responded to both surveys. These small sample sizes contributed to large p-values. Also, the selection of decision-makers was based on identifying a target audience for the initiative and counterparts in the comparison counties. Those individuals who completed both the pre- and post-initiative surveys could have introduced sampling bias.

Fourth, many of the initiative's impacts will take time to emerge and may be subtle, involving shifts in perspective and policy dialogue with respect to nature. Actual policy decisions and other potential ecological impacts are expected to occur beyond the timeframe of our study. A fruitful strategy for supporting a perspective shift over time would be to build connections with local schools systems to bring thinking about ES to younger generations.

Our qualitative results rule out the possibility that the initiative had no impact on decision-makers. The statistical limitations of our sample sizes and spillover between counties are the most likely reasons for non-significant quantitative results. The heterogeneous impact across outcomes and longer time frames needed for impact to manifest are also important. As in all such studies, our survey design (i.e. basic choice of outcomes to measure and wording of questions) introduced an additional measurement error. If we used other proxies for decision-making capacity in our survey, we may have found different levels of impacts.

The qualitative results shed light on the initiative's impact. The interviews uncovered barriers to using ES knowledge that are related to decision-makers reporting less capacity to implement policies or plans about ES post-initiative. In the interviews, most decision-makers felt that the methods underpinning the ES valuation reports needed to be vetted and refined before they could be used to enact new policies. A significant challenge lies in how to effectively integrate “new” ways of valuing land into existing decision processes and tools such as cost-benefit analysis. Efforts to implement policies or plans about ES could be further impaired by academic-style assessments that do not match the needs of people involved in policymaking, development review, or project implementation. Lastly, there is a potential mismatch between the scale of county-wide ES assessments and the scale of individual property-level decisions. These issues warrant consideration by those involved in ES assessment or policy. Despite the challenges associated with using ES knowledge, the processes of the county-wide assessments did affect how decision-makers thought about longer-term, regional planning. Decision-makers appreciated having additional ways to communicate with people about the value of conservation.

2.6. Conclusion

The conservation community could benefit from being more rigorous in assessing impact of ES assessments. Our study highlights the importance of mixing quantitative and qualitative methods to uncover a more nuanced picture of impact (Smith et al., 2012). While our study was not amenable to experimental design, we used mixed methods with a comparison group to understand the impact of the initiative. How can we tell if any

observed impacts were due to the initiative? The comparison group allowed us to consider what we would have observed had the initiative not occurred, and we directly asked decision-makers in interviews to comment on the hypothetical situation where no ES assessments had occurred. By estimating counterfactual situations with a comparison group and through interviews, we can tentatively link observed changes in decision-makers to the initiative.

There are a variety of ways to do impact evaluation. Counterfactual thinking that considers the hypothetical situation in which the treatment is absent, and is an important part of considering rival explanations for observed outcomes (Ferraro & Hanauer, 2014a). Difference-in-differences is a good approach in that it includes quantitative comparison between a treatment group and an estimate of the counterfactual. It does, however, rest on the assumption of equal trends in outcomes among all counties in the absence of the initiative, an assumption that could be tested with surveys at multiple pre-initiative time periods. There are stronger evaluation designs other than difference-in-differences, but we were limited by poor data on observable outcomes and unobservable sources of bias, which is often the case in environmental evaluation (Ferraro & Miranda, 2014). The limited data available means that our findings are best interpreted as suggestive and useful for guiding future studies.

Future research would benefit from longer-term monitoring to evaluate links between various interventions and the subsequent impacts. This would contribute to understanding the full chain of impact from scientific analyses and the knowledge they generate, to changes in the understanding and attitudes of decision-makers, altered policies or plans, and ultimately actual improved outcomes for ES and human well-being.

Multi-site evaluations with better estimates of counterfactuals (perhaps by statistical matching on observable characteristics) and randomized study designs are not always feasible, but should be the aim. At the very least, ES projects need to include impact evaluation as a core component, as the Healthy Lands, Healthy Economies Initiative has.

To move forward and more effectively incorporate ES into decision-making, the planning stages of ES projects need to clearly measure baseline conditions and identify opportunities for knowledge use in collaboration with decision-makers (Waite et al., 2014). During projects, it is important to track how ES knowledge is produced and used. After projects, systematic evaluation of impact and knowledge use over different timescales can provide evidence for what works. This would improve our understanding of how knowledge links to impact and benefit the design of future ES conservation programs.

2.7. References

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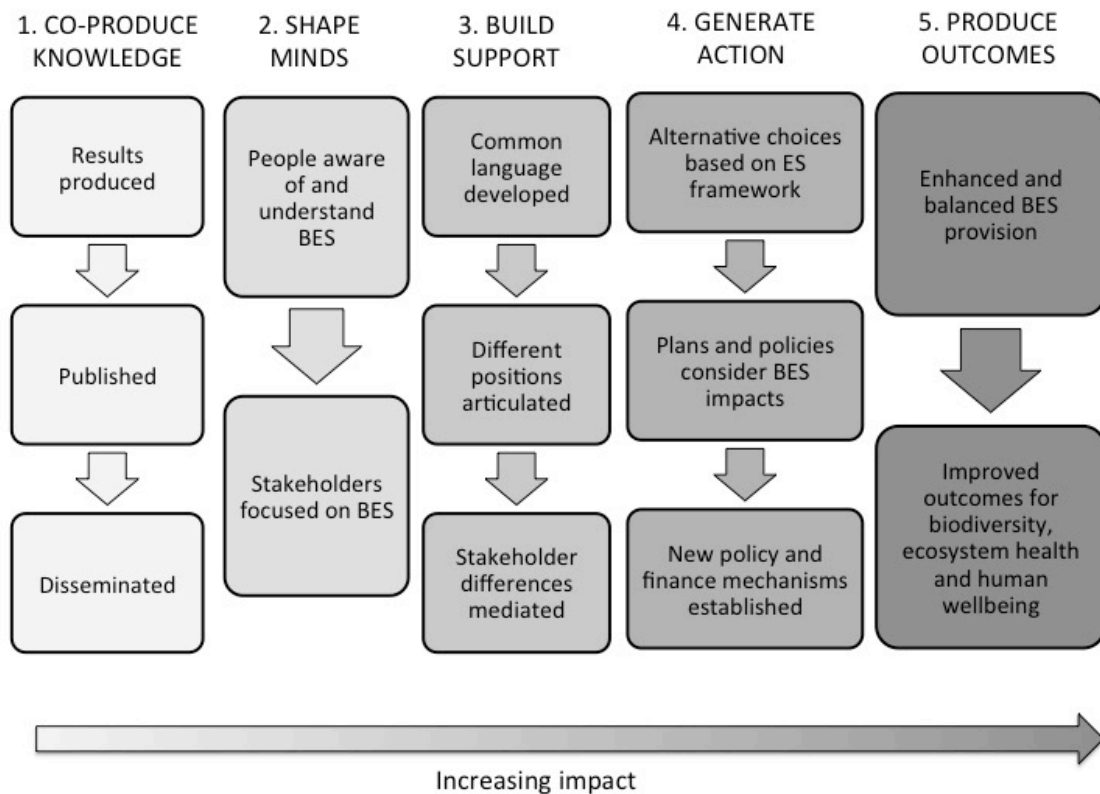


Figure 2.1: Framework for how ES knowledge leads to impact. Five different pathways to impact are represented as columns with increasing impact the further one moves to the right. Our study focuses mainly on pathways 2 and 3. Based on Ruckelshaus et al. (2013) and modified by Posner et al. (in review).

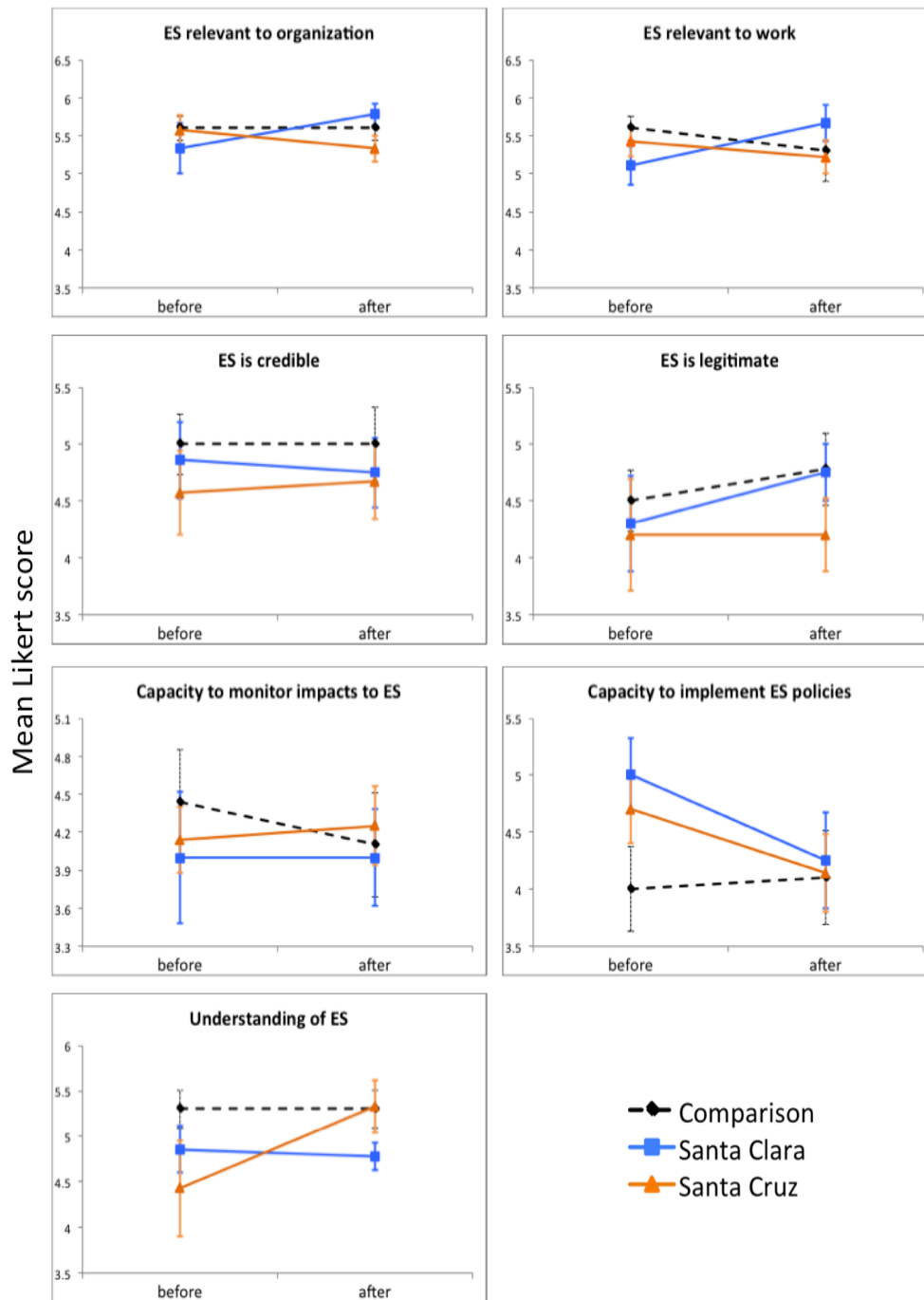


Figure 2.2: Differences in outcomes between the comparison group and the two treatment groups of Santa Clara and Santa Cruz Counties. Y-axis is mean Likert score. Data points are for mean outcomes with standard errors before and after the initiative for the three groups.

Table 2.1: The difference-in-differences method. The two treatment groups of Santa Clara and Santa Cruz Counties had the ES initiative, whereas the comparison group composed of individuals from nearby counties did not have a county-wide ES assessment. $Y_{C, \text{pre}}$ refers to an outcome variable for the comparison group before the initiative.

	Comparison	Santa Clara or Santa Cruz	
Pre	$Y_{C, \text{pre}}$	$Y_{SC, \text{pre}}$	
Post	$Y_{C, \text{post}}$	$Y_{SC, \text{post}}$	
Difference	$Y_{C, \text{post}} - Y_{C, \text{pre}}$	$Y_{SC, \text{post}} - Y_{SC, \text{pre}}$	$(Y_{C, \text{post}} - Y_{C, \text{pre}}) - (Y_{SC, \text{post}} - Y_{SC, \text{pre}})$ = estimate of treatment effect

Table 2.2: Difference-in-differences estimates of the impact of the initiative on decision-maker outcomes. β_3 is the coefficient of the interaction term from our regression models.

Outcomes	Estimate (β_3)	SE	p-value
<i>Santa Clara</i>			
Relevance to org	0.44	0.42	0.30
Relevance to work			
*	0.86	0.57	0.14
Credibility of ES	-0.11	0.63	0.87
Legitimacy of ES	0.19	0.64	0.77
Understanding ES	-0.08	0.42	0.85
Capacity to monitor impacts to ES	0.34	0.87	0.70
Capacity to implement policies	-0.85	0.77	0.28
<i>Santa Cruz</i>			
Relevance to org	-0.24	0.35	0.50
Relevance to work	0.09	0.56	0.87
Credibility of ES	0.10	0.66	0.89
Legitimacy of ES	-0.26	0.70	0.72
Understanding ES*	0.90	0.61	0.15
Capacity to monitor impacts to ES	0.45	0.75	0.55
Capacity to implement policies	-0.53	0.76	0.49

Table 2.3: Percentages of respondents who agreed or strongly agreed with statements about ES in our survey questions.

	Santa Clara pre	Santa Clara post	Santa Cruz pre	Santa Cruz post	Comparison pre	Comparison post
Relevance to org	88.9%	100.0%	77.8%	100.0%	100.0%	100.0%
Relevance to work*	77.8%	88.9%	77.8%	88.9%	100.0%	90.0%
Credibility of ES	66.7%	66.7%	44.4%	55.6%	60.0%	70.0%
Legitimacy of ES	33.3%	55.6%	33.3%	33.3%	30.0%	60.0%
Understanding ES*	55.6%	77.8%	33.3%	77.8%	90.0%	90.0%
Capacity to monitor impacts to ES	33.3%	33.3%	22.2%	44.4%	60.0%	60.0%
Capacity to implement policies**	77.8%	55.6%	55.6%	33.3%	50.0%	40.0%

2.8. Supporting Information

2.8.1. Direct Observation Record Form

Purpose of observations

To provide data for evaluating the impact of this ecosystem services valuation initiative

To understand current issues facing decision-makers and stakeholders

To understand project architecture and processes

Not to report on any individual's or organization's performance

Components to observe

Characteristics of participants

- gender, age, vocation, dress, appearance, ethnicity
- attitude toward subject, others, self
- skill and knowledge level
- statements about commitments, intentions, values, changes to be made

Interactions

- level of participation/interest
- power relationships, decision-making, current issues
- general climate for learning, problem-solving
- levels of support, cooperation

Nonverbal behavior

- facial expressions, gestures, posture
- apparent interest and commitment – impressions of meeting

Leaders/presenters

- clarity of communication and responses to questions
- group leadership skills, encouraging participation
- awareness of group climate and dynamics
- flexibility/adaptability
- knowledge of subject, use of aids, teaching techniques
- sequence of activities

Physical surroundings

Products and outcomes

Questions to consider

How is ecosystem service knowledge presented?

How do participants engage with ecosystem service knowledge?

What relationship do participants have with the scientific knowledge?

What levels of salience, credibility, and legitimacy are present?

Do decision-makers play a role in co-creating ecosystem service knowledge? What kind of a role? What ownership do they have over different aspects of the project?

What are the intended uses of the information or project outputs?

What are major differences and similarities among the counties?

How do stakeholders, meeting participants, and Earth Economics team marshal evidence in making their cases?

2.8.2. Interview Questions

1. I would like to get an idea of how you are involved with the HLHE initiative.
Describe in what capacity you know about the initiative and how your work is related.
2. Do you work directly with anyone involved in the HLHE initiative?
3. How are decisions about conservation and stewardship made in your organization?
Are any decisions made related to Ecosystem Services (ES)?
4. What are the main challenges to incorporating ES into decision-making? How could they be overcome?
5. What questions do you think the Nature's Value report sought to answer?
6. How do you expect the results of the study (such as the information presented in the Nature's Value reports) to be used?
 - a. Describe any particular decisions you think can be informed by the study.
 - b. Describe any particular people or groups you think will use the study.(for example, hazard mitigation, public goods charges, damage assessment, avoided cost estimates)
7. What other influence do you think this ES knowledge has had or will have?

- a. Will it help decide among options (i.e. desal plant or not)?
 - b. Will it illustrate the value of particular options?
 - c. Will it bring new information or knowledge to stakeholders, advisors, or policy makers?
 - d. Will the SC3 project influence the capacity to monitor impacts or implement policies specific to ES? (in Santa Cruz, example of potential use is flood management)
8. Thinking about key future projects/decisions that could be affected by this project...
- a. In Santa Clara: do you think the report on The Value of Nature in Santa Clara County affected your vote or other peoples' votes for Measure Q?
 - b. In Santa Cruz: do you think the report on The Value of Nature in Santa Cruz County could have future uses (such as to support a local funding measure for protection of ecosystem services)?
 - c. In Santa Cruz: do you think the report will influence discussions about the potential formation of a new open space district? If so, how?
9. In terms of the messaging used in the HLHE Initiative...
- a. What worked? What resonated with you or other stakeholders?
 - b. What didn't work?
10. What do you think would occur if there were no ES valuation studies in these counties?
11. Is there anything else you would like to add that I haven't asked about?

2.8.3. Ecosystem Services Survey

1. Please tell us how much you agree with the following statements:

Sub-question	Don't Know	Strongly Disagree	Disagree	Neutral	Agree	Strongly Agree
An ecosystem services approach is relevant to my company's/ organization's work.	1	2	3	4	5	6
My company/organization actively considers ecosystem services in making decisions.	1	2	3	4	5	6
The economic value of ecosystem services is relevant to my own work.	1	2	3	4	5	6
Conservation work is central to my company's/organization's core mission.	1	2	3	4	5	6
Regional or multi-county projects are high priority for my company/organization.	1	2	3	4	5	6
Barriers outside of my organization impede multi-jurisdictional collaboration.	1	2	3	4	5	6
My organization is involved in projects with other agencies/ organizations.	1	2	3	4	5	6
My organization should be involved in more projects with other agencies/ organizations.	1	2	3	4	5	6
Capacity to monitor impacts to ecosystem services exists in my county/region.	1	2	3	4	5	6
Capacity to implement policies or plans about ecosystem services exists in my county/region.	1	2	3	4	5	6
Ecosystem service knowledge is legitimate – gathered in a way that is complete, correct, and unbiased.	1	2	3	4	5	6
The economic value of ecosystem services can be quantified in scientifically credible ways.	1	2	3	4	5	6
I have a solid understanding of what the term <i>ecosystem services</i> means.	1	2	3	4	5	6
I have been familiar with <i>ecosystem services</i> language for more than a year.	1	2	3	4	5	6

2. Some people do not think the term *ecosystem services* is effective in non-scientific settings. Which of the following terms do you prefer as an alternative? (check one)

<input type="checkbox"/>	<i>The value of nature</i>
<input type="checkbox"/>	<i>The economic value of conservation</i>
<input type="checkbox"/>	<i>Conservation's benefits</i>
<input type="checkbox"/>	<i>Nature's assets</i>
<input type="checkbox"/>	<i>Natural capital</i>
<input type="checkbox"/>	<i>Nature's benefits</i>
<input type="checkbox"/>	<i>None of the above, I like the term ecosystem services</i>
<input type="checkbox"/>	<i>Other _____</i>

3. Approximately what percentage of your time would you estimate is spent on work and/or projects that interface with ecosystem services?

(Place a mark at the closest point on the following line *or* check the box "I don't know")

0-----20-----30-----40-----50-----60-----70-----80-----90-----100 %

<input type="checkbox"/>	<i>I don't know</i>
--------------------------	---------------------

4. What percentage of your time do you work in the following places?

Next to each option, please estimate (by marking an X) the percentage of your overall work in that county/region.

Santa Cruz 0-----20-----30-----40-----50-----60-----70-----80-----90-----100 %

Santa Clara 0-----20-----30-----40-----50-----60-----70-----80-----90-----100 %

Sonoma 0-----20-----30-----40-----50-----60-----70-----80-----90-----100 %

Regional level 0-----20-----30-----40-----50-----60-----70-----80-----90-----100 %

State level 0-----20-----30-----40-----50-----60-----70-----80-----90-----100 %

5. How would you characterize your relationship to conservation work? (check all that apply)

<input type="checkbox"/>	Providing scientific or technical input
<input type="checkbox"/>	Advocating for conservation
<input type="checkbox"/>	Mediating relationships

Implementing projects
 Strategic planning
 Making decisions about funding priorities
 Developing regulations or crafting policy
 Enforcing regulations
 Assessing or mitigating impacts
 Other _____

6. Approximately what percentage of your time would you estimate is spent on conservation?

(Place a mark at the closest point on the following line.)

0-----20-----30-----40-----50-----60-----70-----80-----90-----100 %

7. Approximately what percentage of your time would you estimate is spent on management of natural resources?

(Place a mark at the closest point on the following line.)

0-----20-----30-----40-----50-----60-----70-----80-----90-----100 %

8. How important CURRENTLY are the following ecosystems services in your organization's work? Please circle one number for each statement, indicating how important each is to your organization, with 1 being "not important at all", and 5 being "highly important." If you do not know, please circle "0" for "don't know" (*circle one number for each*):

<i>Ecosystem Services</i>	Don't Know	Not important at all	Somewhat unimportant	Neither important or unimportant	Somewhat important	Highly important
Regulation of greenhouse gases (forests store carbon)	0	1	2	3	4	5
Protection from natural disasters (coastal ecosystems mitigate hazards from storms or severe weather)	0	1	2	3	4	5
Flood control	0	1	2	3	4	5
Water quality protection	0	1	2	3	4	5
Soil retention	0	1	2	3	4	5
Soil formation	0	1	2	3	4	5
Nutrient cycling (promotes healthy and productive soils)	0	1	2	3	4	5
Pollination	0	1	2	3	4	5
Biological control (pest and disease control)	0	1	2	3	4	5
Habitat and biodiversity	0	1	2	3	4	5
Food & Agriculture	0	1	2	3	4	5
Timber/forest products	0	1	2	3	4	5
Aesthetic quality (enjoyment of scenery)	0	1	2	3	4	5
Recreation & Tourism	0	1	2	3	4	5
Public Health	0	1	2	3	4	5
Cultural and historic value	0	1	2	3	4	5
Spiritual value	0	1	2	3	4	5
Science and education	0	1	2	3	4	5

9. How important do you think the following ecosystems services SHOULD BE in your organization's work? Please circle one number for each statement, indicating how important you think each service should be to your organization, with 1 being "not important at all", and 5 being "highly important." If you do not know, please circle "0" for "don't know" (*circle one number for each*):

<i>Ecosystem Services</i>	Don't Know	Not important at all	Somewhat unimportant	Neither important or unimportant	Somewhat important	Highly important
Regulation of greenhouse gases (forests store carbon)	0	1	2	3	4	5
Protection from natural disasters (coastal ecosystems mitigate hazards from storms or severe weather)	0	1	2	3	4	5
Flood control	0	1	2	3	4	5
Water quality protection	0	1	2	3	4	5
Soil retention	0	1	2	3	4	5
Soil formation	0	1	2	3	4	5
Nutrient cycling (promotes healthy and productive soils)	0	1	2	3	4	5
Pollination	0	1	2	3	4	5
Biological control (pest and disease control)	0	1	2	3	4	5
Habitat and biodiversity	0	1	2	3	4	5
Food & Agriculture	0	1	2	3	4	5
Timber/forest products	0	1	2	3	4	5
Aesthetic quality (enjoyment of scenery)	0	1	2	3	4	5
Recreation & Tourism	0	1	2	3	4	5
Public Health	0	1	2	3	4	5
Cultural and historic value	0	1	2	3	4	5
Spiritual value	0	1	2	3	4	5
Science and education	0	1	2	3	4	5

10. What are three ecosystem services related to your work, in order of importance?

- 1) _____
- 2) _____
- 3) _____

10b. Please describe briefly how you would go about estimating the economic value of #1 (ecosystem service) that you have listed above?

11. What is your job title?

12. What type of organization to your work for? (please check one box below)

<input type="checkbox"/>	<i>Resource Conservation District</i>	<input type="checkbox"/>	<i>Parks and Recreation Department</i>
<input type="checkbox"/>	<i>Open Space Authority</i>	<input type="checkbox"/>	<i>Special District</i>
<input type="checkbox"/>	<i>State Regulatory Agency</i>	<input type="checkbox"/>	<i>Consulting Company</i>
<input type="checkbox"/>	<i>Water Agency/District</i>	<input type="checkbox"/>	<i>Private Non-Profit/NGO</i>
<input type="checkbox"/>	<i>Funding organization</i>	<input type="checkbox"/>	<i>Private Business</i>
<input type="checkbox"/>	<i>OTHER</i> _____		

12. How old are you (please check one)

<input type="checkbox"/>	21-30	<input type="checkbox"/>	61-70
<input type="checkbox"/>	31-40	<input type="checkbox"/>	71-80
<input type="checkbox"/>	41-50	<input type="checkbox"/>	81+
<input type="checkbox"/>	51-60		

13. Are you ... ? (please check one)

<input type="checkbox"/>	Male
<input type="checkbox"/>	Female

14. What is your highest level of education?

<input type="checkbox"/>	Some school	<input type="checkbox"/>	Bachelor's degree or equivalent
<input type="checkbox"/>	High School	<input type="checkbox"/>	Master's Degree
<input type="checkbox"/>	Some College	<input type="checkbox"/>	Ph.D., M.D., J.D., or equivalent

15. What is your name?*

* Please also note an email address or a phone number if you'd like to talk with us more in-depth about these issues.

CHAPTER 3: WHAT EXPLAINS THE IMPACT OF ECOSYSTEM SERVICES KNOWLEDGE ON DECISIONS?

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3.1. Abstract

Research about ecosystem services (ES) often aims to generate knowledge that influences policies and institutions for conservation and human development. Yet, we have limited understanding of how decision-makers use ES knowledge, or what factors facilitate use. Here we address this gap and report on the first quantitative analysis of the factors and conditions that explain the policy impact of ES knowledge. We analyze a global sample of cases where similar ES knowledge was generated and applied to decision-making. We first test whether attributes of ES knowledge itself predict different measures of impact on decisions. We find that legitimacy of knowledge is more often associated with impact than either the credibility or salience of the knowledge. We also examine whether explanatory variables related to the science-to-policy process and the contextual conditions of a case are significant in predicting impact. Our findings indicate that while many factors are important, attributes of the knowledge best explain the impact of ES science on decision-making. Our results are consistent with both theory and

previous qualitative assessments in suggesting that the attributes and perceptions of scientific knowledge are important determinants of whether that knowledge leads to action.

3.2. Significance Statement

Our study introduces a conceptual framework and empirical approach to evaluate the impact of scientific knowledge on decisions. We illustrate this novel approach with a sample of 15 international cases involving ecosystem services research, but it has broad applicability. Our results demonstrate that the perception of research knowledge as legitimate (unbiased and representative of multiple points of view) is of paramount importance for impact. More surprisingly, we found that credibility of knowledge is not a significant factor for impact. To enhance the legitimacy needed for knowledge to stimulate action, ES researchers must engage meaningfully with stakeholders to incorporate diverse perspectives transparently. Our results indicate how research can be designed and carried out to maximize the potential impact on real-world decisions.

3.3. Introduction

3.3.1. Ecosystem Services Knowledge Use In Decision-Making

The ongoing loss of biological diversity and persistence of poverty have sparked interest in policies that protect, restore, and enhance ecosystem services (ES). In response, there has been a growth in ecosystem service research that aims to inform policies, incentives, and institutions on a large scale (Daily & Matson, 2008; Daily et al., 2009; Hogan et al., 2011). Despite this goal, scientific knowledge about ES continues to

have limited impact on policy and decisions (Daily & Matson, 2008; de Groot et al., 2010; Laurans et al., 2013; Liu et al., 2010; Spilsbury & Nasi, 2006).

The fact that most land and resource use policy decisions still do not take ES into account stems in part from an ineffective or non-existent interface between ES science and policy, a lack of attention to decision-making processes, and challenges in clarifying conflicting stakeholder values (Eppink et al., 2012; Knight et al., 2008; Mermet et al., 2014; Nesshover et al., 2013). The ES research and policy communities are too often disconnected from one another, with limited interactions, infrequent exchanges of information, and different objectives that hinder coordinated science and policy processes (Weichselgartner & Kasperson, 2010). Many scientists conduct ES research without fully considering how the knowledge they are producing might be used (Laurans et al., 2013). If we want ES information to be incorporated into decisions, then we need to understand how and why decision-makers use certain kinds of information.

Much of the evidence for how and why ES knowledge influences policy decisions is anecdotal. A few recent studies have focused on this issue with qualitative, in-depth case studies (Laurans & Mermet, 2014; MacDonald et al., 2014; McKenzie et al., 2014). To more generally understand this issue, however, we also need quantitative, empirical research into how and why ES knowledge has an impact on decisions (Laurans & Mermet, 2014; Laurans et al., 2013; Liu et al., 2010; McKenzie, et al., 2014). This is an understudied area of research, not least because empirical data on impacts from replicate cases are difficult to compile.

Here we report on a quantitative approach to understanding the factors and conditions that enable ES knowledge impact in decision-making. More carefully

examining the relationships between impacts and enabling conditions helps us better understand why an ES approach may generate impacts on decisions and why it may not.

3.3.2. Enabling Conditions Framework

Understanding the factors that explain impact will benefit those who produce ES knowledge (i.e. by illuminating effective strategies for enhancing knowledge use) as well as decision-makers (i.e. by encouraging their participation in defining use-inspired science). Cash et al. (2003) identify salience, credibility, and legitimacy of knowledge as important enabling conditions for linking sustainability knowledge to action. Salience refers to the relevance of scientific knowledge to the needs of decision-makers, credibility comes from scientific and technical arguments being trustworthy and expert-based, and legitimacy refers to knowledge that is produced in an unbiased way and that fairly considers stakeholders' different points of view. Their framework has inspired others to investigate these three attributes and how they affect decision-makers using knowledge (Cook et al., 2013; Keller, 2010; Reid et al., 2009; Rowe & Lee, 2012; Sarkki et al., 2013).

Others focus on process rather than content of environmental management and policy (Andrews, 2002; Cox & Searle, 2009; Karl et al., 2007; Rosenthal et al., 2014). They describe the importance of joint fact-finding and iterative processes of engagement among scientists and policymakers. Another branch of research has focused on more contextual conditions about the institutions, governance, and culture of places where environmental policy is successful. Haas et al. (1993) and Wunder et al. (2008) note institutional capacity to monitor environmental conditions and enforce rules as critical to effective science-based policies.

We organize these perspectives into three categories of enabling conditions for ES knowledge to lead to action (Table 1). The first category contains variables related to attributes of the scientific knowledge produced. Variables in the second category focus on characteristics of the process through which science informs decisions and policy. The third category contains variables that reflect contextual conditions of the project or place in which it is located.

3.3.3. Research Question And Hypotheses

Drawing from these theoretical frameworks about linking knowledge with action, we test quantitatively which enabling conditions can explain the impact of ES science across a global sample of 15 cases. We analyze this set of science-policy interventions through the attributes of the 1) knowledge produced, 2) science-to-policy process, and 3) contextual conditions of each case. In so doing, we address the question ‘What explains the impact of ecosystem services knowledge on decisions?’ In answering this question, we aim to identify conditions that enable the impact of ES science.

We hypothesize that

- H1: Higher levels of salience, credibility, and legitimacy of ecosystem service knowledge are associated with higher measures of impact.
- H2: Predictor variables related to knowledge are more significant than those related to process or contextual conditions in explaining impact.

3.4. Methods

3.4.1. Sample Of Cases

To examine whether certain factors and conditions predict impact, we sought a sample of cases in which similar scientific tools and approaches were used, but different levels of impact achieved. We used a global sample of case studies from The Natural Capital Project, in which a standardized scientific tool, InVEST, was applied to decisions with the aim of improving conservation, human development and environmental planning outcomes (Table 2). The Natural Capital Project was formed in 2006 by World Wildlife Fund (WWF), The Nature Conservancy, Stanford University, and the University of Minnesota, under the premise that information on biodiversity and ecosystem services can be used to inform decisions that improve human well-being and the condition of ecosystems (Ruckelshaus et al., 2013). InVEST (Integrated Valuation of Environmental Services and Trade-offs) is a suite of software models that can be used to map, quantify and value ecosystem services (Arkema et al., In Press; Bhagabati et al., 2014; Nelson et al., 2009).

3.4.2. Measuring Enabling Conditions (Explanatory Variables)

We sent an electronic survey to decision-makers and boundary organization contacts in each of the demonstration sites presented in Ruckelshaus et al. (2013). Boundary organizations were NGOs that aimed to create more effective policymaking by spanning/bridging the science and policy communities (Guston, 2001). We received survey responses from 15 cases, providing a 40% response rate (Table 2). The survey collected self-reported, ordinal scale data on the variables that we identified in the literature as important elements of an effective science policy interface, or that team members of the Natural Capital Project proposed as conditions that increase the likelihood of knowledge impact (Table 3). This is not an exhaustive list of all potential

enabling conditions, but includes variables that have been identified by multiple sources as significant. Specific survey questions are included in supplementary information.

3.4.3. Measuring Impact (Outcome Variables)

We use the conceptual framework for impact described by Ruckelshaus et al. (2013). We modified the framework to include five pathways through which impact is achieved in ES projects (Figure 1). We added a fifth pathway to reflect recent insights into ways to define ES knowledge impact to include co-production of knowledge (Pathway 1), conceptual use (Pathway 2), strategic use (Pathway 3), instrumental use (Pathway 4), and outcomes for human wellbeing, biodiversity and ecosystems (Pathway 5) (McKenzie et al., 2014). We designed a scoring rubric with a 5-point scale based on this evaluative framework.

Three reviewers (one of whom is a co-author) analyzed a qualitative review of the impacts in each case from Ruckelshaus et al. (2013), written documentation of the cases (including project reports, management plans, or case study summaries), and online resources pertaining to the cases (such as project websites or presentations to decision-makers). The reviewers then provided initial impact scores for each case. Through a Delphi process, the reviewers then gathered and discussed results before independently revising their scores. We averaged the three reviewer scores to obtain, for each case, an estimate of impact 3 (build support), impact 4a (generate action: proposed plans and policies based on ES), and impact 4b (generate action: new policy or finance mechanisms for ES established). Similar methods have been described by Sutherland (2003) and used by Sutherland et al. (2011) and Kenward et al. (2011) to evaluate the impact of science and governance strategies.

A fourth measure of impact included in our analysis, impact 2 (awareness and understanding of ES) was based on survey questions to decision-makers rather than the expert scores (Table 4). We omitted columns 1 and 5 of the evaluative framework from this study because all cases involved co-producing and publishing research results, and it is difficult to show whether biodiversity, wellbeing or ES outcomes were enhanced over the timescale of these projects (Figure 1). Given the post facto nature of this research, the design of the study is limited by the inability to use rigorous impact evaluation methods (Ferraro et al., 2012; Gertler et al., 2010; Margoluis et al., 2009).

3.4.4. Analyses

Inter-rater reliability analysis was used to compare among the three expert reviewers who measured impact in the cases. We calculated a Krippendorff's alpha to measure agreement for ordinal data among three reviewers. From our sample, $\alpha = 0.0544$ for Impact 3, $\alpha = 0.655$ for Impact 4a, and $\alpha = 0.619$ for Impact 4b. For conclusions based on the impact data, we took $\alpha > 0.6$ as acceptable for this study (Krippendorff, 2004). The low level of agreement among reviewers for Impact 3 indicates that only tentative conclusions should be drawn from these data.

We treated the 15 cases as independent data points in the analysis, each with 4 outcome variables (i.e., measures of impact; Table 4) and 16 explanatory variables (i.e., enabling conditions; Table 3). To test H1, we used the 5-point scale to group cases into 3 broader categories (low, medium, high) for salience, credibility, and legitimacy. For example, the lowest levels of credibility of ES knowledge were labeled as "low," the middle two self-reported levels were labeled "medium," and the highest level assigned to the cases by survey respondents was labeled "high." We then used analysis of variance

(ANOVA) to test whether higher levels of salience, credibility, and legitimacy are associated with higher impact for all four impact measures. The unequal number of cases in each of the groups led to an unbalanced ANOVA design.

To test H2 about which enabling conditions could best explain impact, we used an information theoretic approach (Kenward et al., 2011). We first reduced the dataset by using principal components analysis (PCA) on explanatory variables with a Spearman rank correlation coefficient > 0.80 (see Supplementary Information for PCA plots) (Dormann et al., 2013). Using the first principal component for groups of highly correlated variables allowed us to focus on 12 explanatory variables. We used the R package MuMIn to conduct multi-model inference (Barton, 2014). We tested all possible linear models with these 12 variables, for each of the 4 measures of impact. To determine which predictors best explain variability in impact, we ranked the top models by AICc values and calculated model average coefficients with 95% confidence intervals for each of the predictors. Model average coefficients represent the average coefficient for each explanatory variable across all models, weighted for goodness of fit of the models (Burnham & Anderson, 2002).

3.5. Results

For almost all impact measures, we find that impact tends to increase with higher levels of credibility, salience, and legitimacy (Figure 2). With legitimacy, this effect is significant for three of the four measures of impact; with salience, two measures; and with credibility, none (Figure 2; Table 5).

We also find that certain attributes of the ecosystem service knowledge explain impact better than characteristics of the process or contextual conditions (Figure 3). Again, legitimacy emerges as a strong predictor of impact; averaging coefficients across all possible models, we find that legitimacy of the ecosystem service knowledge is better at explaining impact than any other included variable. For all measures of impact, the top models include legitimacy as the strongest variable for explaining impact (see Supplementary Information for table of model selection results with AICc values for top models). Other variables included in best models (ranked by lowest AICc value) are the number of interactions between scientists and decision-makers, the institutional capacities in the cases, and the degree to which local knowledge was incorporated into decisions.

3.6. Discussion and Conclusion

We develop a quantitative approach to examine the conditions under which scientific knowledge about ecosystem services most influences policies and decisions. Using four measures of impact in a global sample of ES cases, we find that legitimacy of scientific knowledge explains impact more than any of the other explanatory variables we tested. Interestingly, higher levels of credibility are not associated with higher levels of impact, however measured. Credibility, or the trustworthy and expert base of the scientific information, is the factor that scientists are most responsible for, while salience and legitimacy are established by both scientists and decision-makers (Rowe & Lee, 2012). The scientific adequacy of ES knowledge is undoubtedly important, perhaps as a necessary precondition to policy processes, but this study finds that it is not significantly associated with higher levels of impact.

The finding that legitimacy appears to matter more than credibility puts great responsibility on researchers to engage with stakeholders. Researchers need to do more to make the knowledge they produce legitimate, and research institutions need to put in place the incentives and time required for researchers to do this.

We also find that different factors are important for different stages of impact (Figure 3). Evidence from practitioners in the field and qualitative studies claim that salience, credibility, and legitimacy are important to generate policy action, even while they recognize tradeoffs may be necessary among these attributes (Cash et al., 2003; Keller, 2010; Reid et al., 2009; Sarkki et al., 2013). Our results indicate that these attributes are not equally important for each stage of impact we considered.

Salience appears to be important at early stages of the policy process, in shaping people's ideas and discussions about ES (Figure 2). Knowledge perceived to be salient – relevant to the needs of decision-makers – is more likely to increase awareness. Greater perceived legitimacy of ES knowledge is significantly associated with greater impact for three measures of impact, including changing awareness, building support, and drafting plans and policies that consider ES (Figure 2). This reinforces the idea that it is important for decision-makers to view ES knowledge as unbiased, and based on a fair consideration of different stakeholder values at all stages of decision-making. Transparently incorporating key diverse perspectives surrounding an issue can build trust and improve decision-makers' acceptance of knowledge as legitimate (Young et al., 2013).

Regular interactions between scientists and decision-makers are important for impact 3 (building support). Building support is a political process of aligning shared interests behind particular positions, and interactions among decision-makers and

stakeholders also take place at many of those same events/meetings. Decision-makers are likely to perceive the resulting scientific knowledge as salient when relevant policy questions, which they help frame, inspire science. Scenarios of future conditions and collaborative processes among scientists and decision-makers can also ensure the salience and legitimacy of knowledge-producing efforts (Rosenthal et al., 2014).

Interestingly, local knowledge and institutional capacities are also important, but with negative coefficients, indicating an inverse relationship with this measure of impact. This is due to a few cases where low impact was achieved despite local knowledge being included, and where high impact was achieved despite low institutional capacities. Scientists and decision-maker interactions, local knowledge, and institutional capacities are also important for explaining impact 4a (draft plans and policies consider ES) and impact 4b (new plans and policies for ES are established).

Viewed as a whole, these results indicate that early on in a science-to-policy cycle, the perception of ES knowledge as legitimate and salient is important to help shape conversations and raise awareness. Later on in the science-to-policy cycle, the contextual conditions that are outside of scientists' control gain importance. According to these findings, the factors that best predict the final stage of impact (when a project results in draft or established policies that consider ES) are the degree to which decision-makers perceive ES knowledge as legitimate, the institutional capacities in the place where the project occurs, the use of local knowledge, and the amount of interaction between scientists and decision-makers (Figure 3 and Table S3). However, there could be interactions among these variables or effects that our analysis did not uncover because of our sample size (for example, legitimacy only matters when credibility is high). And, in

the final stages of a policy process when a new policy or finance mechanism actually becomes established, there are a multitude of variables at play, including many not measured here.

While our study illustrates a potentially powerful empirical approach to these issues, several limitations should be kept in mind. First, in exploring these relationships, it is difficult to link a policy change to any specific causal factor, because so many competing variables influence policy development (Ferraro et al., 2012). Studies with multiple case study comparisons complement our results by taking into account many of the subtle issues at play within the context of each unique case (Creswell, 2009; McKenzie et al., 2014; Yin, 2009). Second, a sample of only 15 cases limits our statistical power and ability to infer general relationships. Nevertheless, consistent trends observed across several impact measures (e.g., Figure 2) instill some confidence in our overarching results. Assembling larger datasets and measuring impact with a standard framework across researchers will allow future studies to strengthen confidence in general findings. Third, expert opinion carries inherent potential for bias and error, but is increasingly well understood and supported as an empirical approach for research at the intersection of science and policy (Manos & Papathanasiou, 2008). While observer bias remains an issue, the relative differences observed among cases are more robust.

Despite these limitations, our study advances our understanding of enabling conditions, use of ecosystem service knowledge, and the elements that lead to an effective science-policy interface (Kenward et al., 2011; Mermet et al., 2014; Waite et al., 2014). Understanding the factors that tend to enhance the policy impact of ES knowledge is critical. Unless we consider the relationships that decision-makers have with the

products and process of science, the impact of ES knowledge will be haphazard, and will not prevent the continued declines in ecosystems, biodiversity, and the benefits they provide to people.

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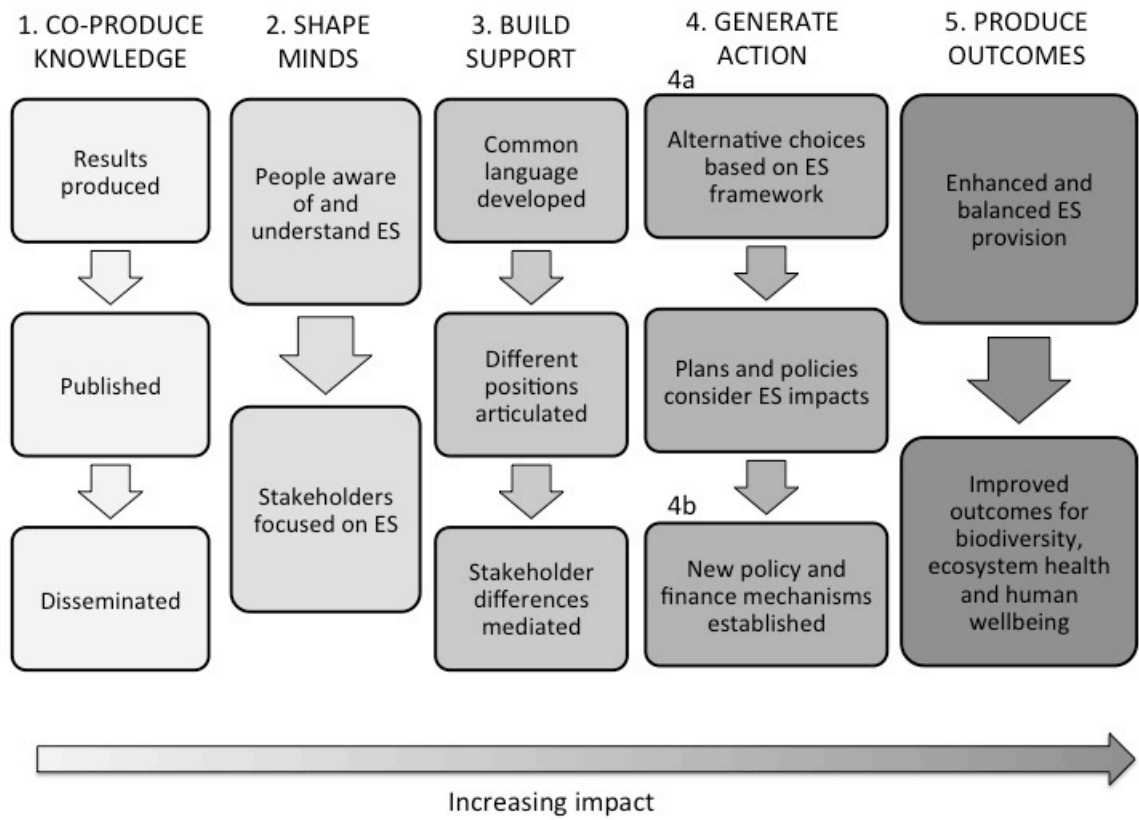


Figure 3.1: Evaluative framework for how ES knowledge leads to impact. Each column represents a pathway to different forms of impact, with increasing levels of impact going from left to right. Columns 2, 3, and 4 (a and b) were the basis for our measurement of impact in each of the cases.

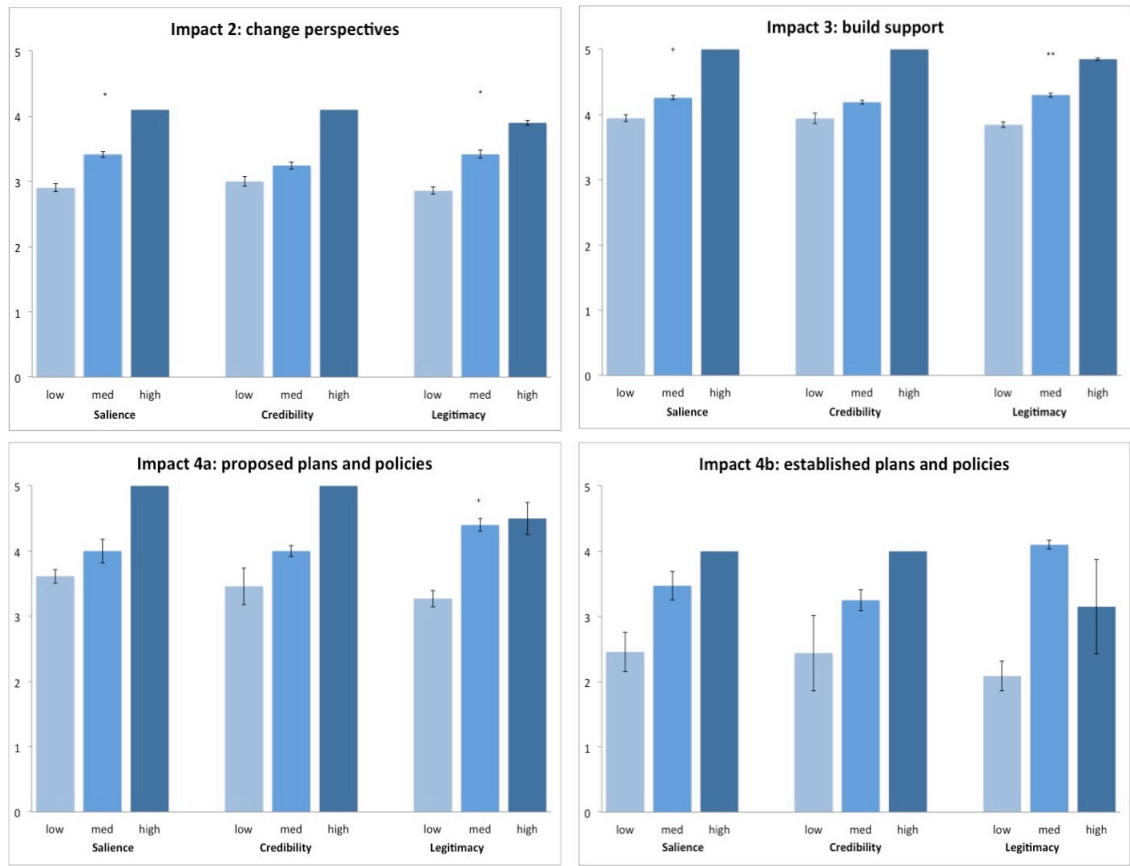


Figure 3.2: Effects of knowledge attributes on policy impact. Bars depict mean levels of impact for different levels of salience, credibility, and legitimacy in the 15 cases. No standard error bar indicates 1 case in that category. ** $p < 0.01$; * $p < 0.05$; + $p < 0.1$

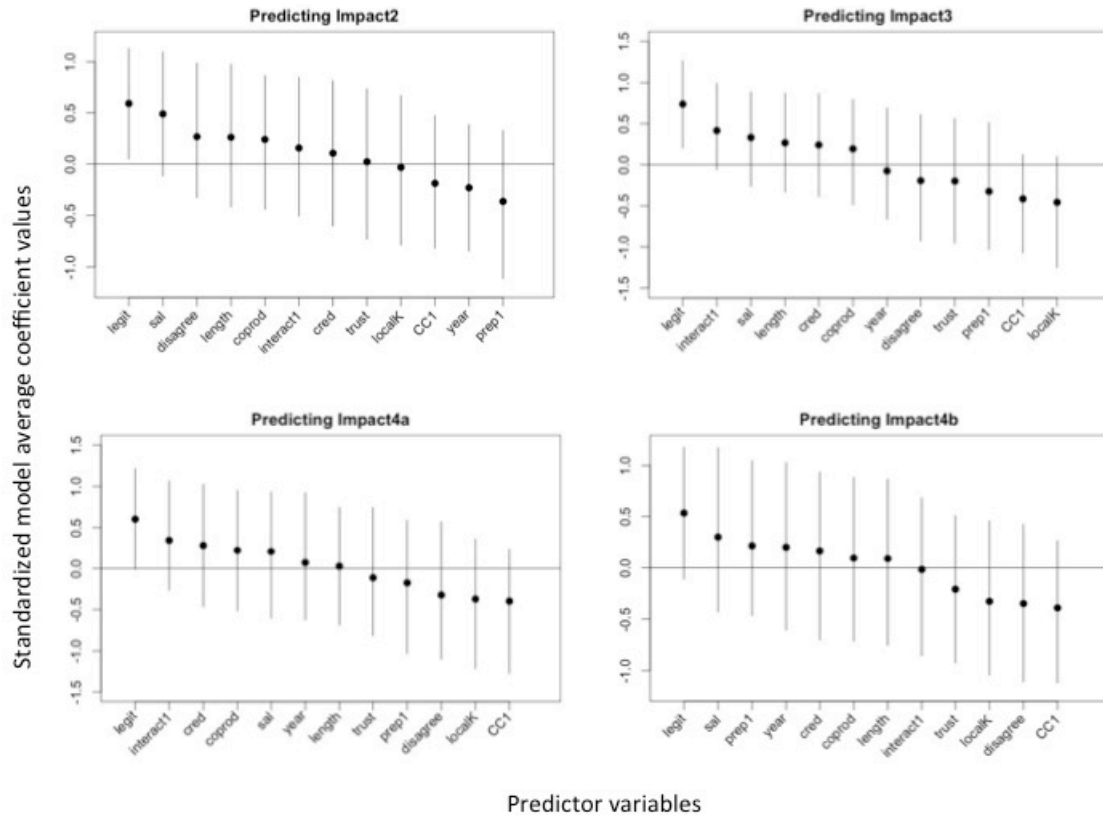


Figure 3.3: Effects of multiple attributes on policy impact. Points and whiskers represent model average coefficients with 95% confidence intervals. Predictor variables are explained in Table 3. The first principal components were used for three groups of highly correlated predictors (Spearman rank correlation coefficient ≥ 0.80): Interact1 (level of joint production as estimated by interactions in person or by phone/email), prep1 (decision-making power and stakeholder representation), and CC1 (institutional capacities to measure baseline ES and human activities, monitor changes to ES and human activities, and implement policy).

Table 3.1: Enabling conditions that facilitate the success of ecosystem service projects, as suggested by qualitative reviews of projects.

Reference	Attributes of knowledge	Process	Contextual conditions
Cox and Searle (2009)	Clear science about ES, interactions between services, and how proposed actions may affect services	A confined system with clearly identified stewards, perpetrators of negative impacts, and service beneficiaries	Good governance in terms of clearly defined ownership or tenure, a legal system, capacity to enforce laws and monitor impacts, and a functioning infrastructure to support projects
Waite et al. (2014)	A clear policy question; A clear presentation of methods, assumptions, and limitations	Strong stakeholder engagement; Effective communications and access to decision-makers	Good governance; Local demand for valuation; Economic dependence on resources; High levels of threats to coastal resources
Rosenthal et al. (2014); McKenzie et al. (2014); Ruckelshaus et al. (2013)	Policy question; Pertinent data; Integration of local and traditional knowledge	Meaningful participation and engagement with diverse groups; Joint knowledge production; Iterative process; Scenario development	Capacity to measure ES; Established planning process; Policy window

Table 3.2: The sample of 15 global cases in which InVEST was used in a policy decision

context. Each case represents a data point in the analysis. Ruckelshaus et al. (2013) discusses cases in more detail.

	Location	Decision Context	Organizations of survey respondents
1	Belize	Spatial planning	Coastal Zone Management Authority and Institute (national government)
2	Canada - West Coast Vancouver Island	Spatial planning	West Coast Aquatic (board with representation from local and provincial government, nine First Nations, conservation NGOs, and businesses)
3	Colombia - Cauca Valley Water Fund	Water Funds	Cauca Valley Water Fund, The Nature Conservancy
4	Colombia - Cesar Department	Permitting & mitigation	The Nature Conservancy
5	China - Baoxing County, Hainan Island, Upper Yangtze River Basin	Spatial planning for Ecosystem Function Conservation Areas	Chinese Academy of Sciences, The Natural Capital Project
6	Himalayas (Bhutan, Nepal, India)	Spatial planning	WWF Eastern Himalayas Program
7	Latin America Water Funds Platform	Water funds	The Nature Conservancy
8	Indonesia - Borneo	Spatial planning, policy advocacy	WWF Indonesia
9	Indonesia - Sumatra	Spatial planning, Strategic Environmental Assessment	WWF Indonesia
10	Tanzania - Eastern Arc Mountains	PES & REDD planning, policy advocacy	Valuing the Arc project at Cambridge University and WWF
11	United States – Ft. Lewis-McChord and Ft. Pickett	Spatial planning for military installation activities	US Army and Air Force (Department of Defense)
12	United States - Galveston Bay in Texas	Hazard mangement, spatial planning, climate adaptation	SSPEED (Severe Storm Prediction, Education and Evacuation from Disasters) Center led by Rice University, The Nature Conservancy marine science
13	United States - Monterey Bay in California	Climate adaptation	Santa Cruz County, Moss Landing Marine Labs of California State Universities

14	United States - North Shore of O`ahu in Hawai`i	Spatial planning	Kamehameha Schools (private land-owner)
15	Virungas Landscape - DRC, Uganda, Rwanda	Permitting & mitigation, Strategic Environmental Assessment	Institute of Tropical Forest Conservation

Table 3.3: Summary of predictive factors in three broad categories of enabling conditions.

See s1 for survey questions.

Variable	Description	Survey question	Reference
Attributes of knowledge			
Salience	The relevance of scientific knowledge to the needs of decision-makers	Q1	Cash et al. (2003); Hirsch and Luzadis (2013)
Credibility	How trustworthy and expert-based are the scientific and technical arguments	Q2	Cash et al. (2003)
Legitimacy	Unbiased knowledge that fairly considers stakeholders' different points of view	Q3	Cash et al. (2003)
Traditional knowledge	Whether local knowledge and experience was included in ES assessment and planning	Q4	Watson (2005)
Process			
Level of joint production *	Scientists, stakeholders, and decision-makers working together to produce ES information; amount of interaction by phone/email or in person	Q5, Q6, and Q7 as separate variables	Cash et al. (2003); Karl, Susskind, Wallace (2007); Lee and Rowe (2013)
Stakeholder representation **	Proportion of people impacted by decisions about ES that were included	Q8	Beierle and Konisky (2001); Young et al. (2013); Lee (2003); Reed (2008)
Presence of conflict/consensus	Level of perceived conflict or disagreement among stakeholders	Q9	Karl, Susskind, Wallace (2007)
Trust	Amount that trust between stakeholders increased during project	Q10	Pretty and Ward (2001); Pretty and Smith (2004)
Length	Length, in years, of project		Natural Capital Project
Contextual conditions			
Capacity to measure baseline ES and	Assessed for at the start of the project	Q11	Ferraro, Hanauera, Sims (2011); Smit

human activities ***			and Wandel (2006)
Capacity to monitor changes to ES and human activities ***	Assessed for at the start of the project	Q12	Levy, Keohane, Haas (2001); Smit and Wandel (2006)
Capacity to implement policy ***	Assessed for at the start of the project	Q13	Levy, Keohane, Haas (2001); Smit and Wandel (2006)
Decision-making power **	Concentrated or shared distribution of decision- making power among stakeholders	Q14	Natural Capital Project
Year	Year project began		Natural Capital Project

Note: Multi-model inference was conducted with the following first principal components based on the variables noted in the table: * Interact1; ** Prep1; *** CC1.

Table 3.4: Questions to assess impact (the response variable) according to the evaluative framework.

Impact pathway	Measured with	Questions for survey respondents (for impact 2) or expert reviewers (for impact 3, 4a, and 4b)
2: Change perspectives	Survey	What proportion of stakeholders was aware of and understood ES before the ES project? After the project?
3: Build support	Expert scoring	How much were the science and InVEST results used to build support for ES among stakeholders? Did the ES knowledge help develop a common language among stakeholders, articulate different positions, or mediate differences?
4a: Generate action	Expert scoring	How much did draft plans or policies emerge that consider ES? Did proposed plans or policies consider ES?
4b: Generate action	Expert scoring	Did a plan, policy, or finance mechanism to enhance, conserve, or restore ES become established?

“Stakeholders” refers to people or groups with an interest or concern in policy decisions that affect ES (for example, individual landowners, conservation NGOs, private businesses).

Table 3.5: Relationships between attributes of knowledge and policy impact. Each element reports the $F_{1,15}$ value (and corresponding p-value) for the ANOVA results.

	Impact 2	Impact 3	Impact 4a	Impact 4b
Salience	6.04 (0.029)*	3.23 (0.095)+	1.29 (0.277)	1.29 (0.276)
Credibility	2.14 (0.167)	2.42 (0.144)	1.23 (0.288)	0.36 (0.559)
Legitimacy	8.89 (0.011)*	10.25 (0.007)**	4.60 (0.052)+	2.78 (0.119)

** 0.01; * 0.05; + 0.1.

3.9. Supporting Information

3.9.1. Survey Questions

Q1. How relevant and timely was the ecosystem service information?

Not at all, only a little, somewhat, very, a great deal

Q2. Was the ecosystem service information scientifically credible? (“scientifically credible” refers to whether the information was reliable and based on scientific expertise)

Not at all, only a little, somewhat, very, a great deal

Q3. Did the decision-making process represent many diverse views on the management or policy issues?

Not at all, only a little, somewhat, a lot, a great deal

Q4. How much was local knowledge and experience included... a) in the ecosystem service assessment? b) in management or planning decisions?

Not at all, only a little, somewhat, a lot, a great deal

(responses to a and b were averaged together)

Q5. How much did scientists, stakeholders, and decision-makers work together to co-produce ecosystem service information?

Not at all, only a little, somewhat, a lot, a great deal

Q6. How many time did you interact with scientists by phone or email?

0, 1-5, 6-10, 11-15, 16-20, more than 20 times

Q7. How many time did you interact with scientists in person?

0, 1-5, 6-10, 11-15, 16-20, more than 20 times

Q8. About what proportion of the people impacted by decisions about ecosystem services were represented in the process of... a) producing ecosystem service information? b) using ecosystem service information in decisions?

None at all, some, about half, most, all

(responses to a and b were averaged together)

Q9. How much conflict or disagreement was there among stakeholders during the process of... a) producing ecosystem service information? b) using ecosystem service information in decisions?

None at all, only a little, some, a lot, a great deal

(responses to a and b were averaged together)

Q10. How much did trust between stakeholders increase during the process of... a) producing ecosystem service information? b) using ecosystem service information in decisions?

None at all, only a little, somewhat, a lot, a great deal
(responses to a and b were averaged together)

Q11. Was there institutional capacity to measure baseline ecosystem services and human activities before the project?

None at all, only a little, somewhat, a lot, a great deal

Q12. Was there institutional capacity to monitor changes to ecosystem services and human activities before the project?

None at all, only a little, somewhat, a lot, a great deal

Q13. Was there institutional capacity to implement policies that impact ecosystem services and human activities before the project?

None at all, only a little, somewhat, a lot, a great deal

Q14. How was decision-making power distributed among stakeholders?

Decision-making power very concentrated, somewhat concentrated, evenly concentrated and shared, somewhat shared, decision-making power very shared

Q15. How long was the ecosystem service project?

Less than 1 year, 1 year, 2 years, 3 or more years

Q16. What year was the project completed?

3.9.2. Principal Components Analysis

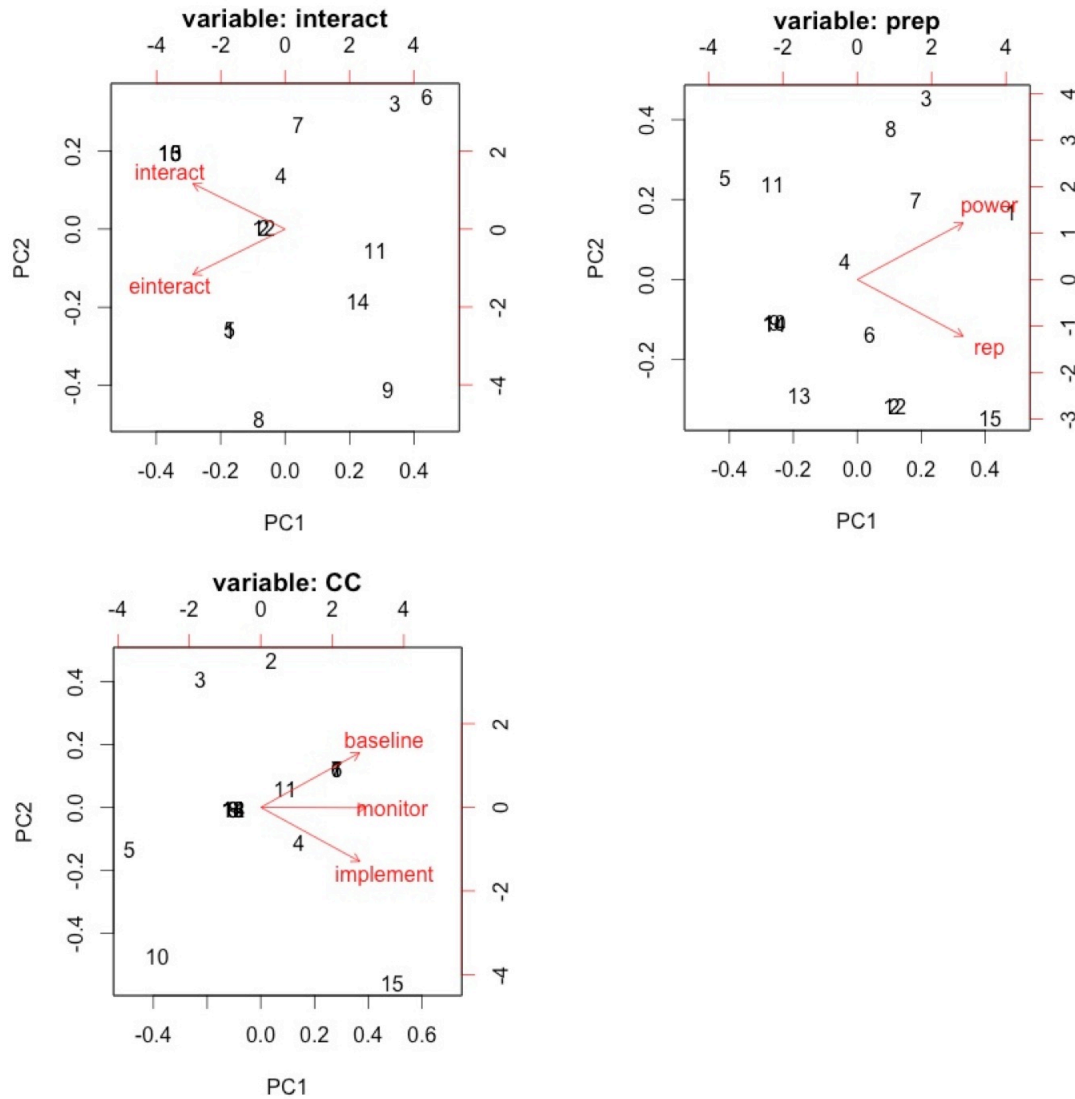


Figure 3.4: PCA bi-plots. We reduced the dataset by combining the following highly correlated variables. prep: power and representation (correlation coefficient = 0.74); interact: in-person interactions and electronic interactions (correlation coefficient = 0.72); CC: contextual conditions as institutional capacity to measure baseline conditions, monitor changes, and implement policies (paired correlation coefficients of 0.70, 0.79, and 0.80). In subsequent analyses, we used PC1 for explanatory variables (or the inverse of PC1, so that increases in the principal component correlated to increases in the underlying variables).

3.9.3. Model Selection

Table 3.6: Model selection results with top 5 models and AICc values. Check mark ✓

indicates variables included in each model.

IMPACT 2						
Model\Var	legitimacy	length	power & representation	salience	-	AICc
1	✓					30.0
2	✓			✓		31.5
3	✓	✓	✓			32.0
4	✓					32.1
5				✓		32.1
IMPACT 3						
Model\Var	legitimacy	local knowledge	interactions	institution capacity	-	AICc
1	✓	✓				22.3
2	✓		✓			22.7
3	✓		✓	✓		22.9
4	✓			✓		23.5
5	✓					24.0
IMPACT 4a						
Model\Var	legitimacy	local knowledge	interactions	institution capacity	-	AICc
1	✓					45.0
2	✓	✓				45.1
3	✓			✓		45.2
4	✓		✓			47.2
5			✓			47.4
IMPACT 4b						
Model\Var	legitimacy	local knowledge	disagreement	institution capacity	salience	AICc
1	✓			✓		56.1
2	✓					56.3
3	✓	✓				57.0
4			✓			57.5
5					✓	57.8

CHAPTER 4: GLOBAL USE OF ECOSYSTEM SERVICE MODELS

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4.1. Abstract

Spatial models of ecosystem services are increasingly important for informing land use and development decisions. Understanding who uses these models and the conditions associated with model use is critical for increasing their impact. We tracked the use of The Natural Capital Project's ecosystem service models (InVEST) over a 25-month period and observed that 19 different models were run 43,363 times in 104 countries. We analyzed the 25,431 models runs that were not tied to model development activity. Models for regulating services (e.g., carbon and sediment retention) were the most commonly used. We analyzed relationships between country-level variables and use of ecosystem service models and found that capacity (population, GDP per capita, Internet and computer access, and InVEST trainings), governance, biodiversity, and conservation spending are positively correlated with use. Measures of civic involvement in conservation, carbon project funding, and forest cover are not correlated with use. Using multivariate statistical models, we analyzed which combinations of country-level variables best explain the use of InVEST and found further evidence that variables

related to capacity are the strongest predictors. Finally, we examined InVEST trainings in detail and found a significant effect of trainings on subsequent use of InVEST models ($p < 0.001$). Our results indicate that the general capacity of a country may limit uptake and use of these decision support tools. Trainings are thus more likely to have a large impact if other capacity aspects are present. Thoughtfully tracking and analyzing the use of decision support models can help us understand the user audience and context, and design better tools.

4.2. Introduction

Ecosystem services, the benefits that people receive from nature, are degraded and projected to decline further over the first half of this century (Millennium Ecosystem Assessment Program, 2005). Since many current policy and economic decisions do not account for the values of ecosystem services (ES), planners and decision-makers are increasingly focused on the management of ES as a viable way to understand and manage human interactions with ecosystems (Braat & Groot, 2012; Holzman, 2012). The concept of ES is becoming essential in many of today's largest conservation organizations and academic research about the topic has increased steadily over the past two decades (Abson et al., 2014; Fisher et al., 2009).

As the ES concept grows more popular, there is more demand for ES information that has the potential to affect policy decisions (Daily & Matson, 2008; Daily et al., 2009; Mermet et al., 2014; Ruckelshaus et al., 2013). In particular, computer models that generate spatially explicit information about ES are commonly used to inform decisions (Burkhard et al., 2013; Crossman et al., 2012). The information these tools produce often

illustrates how landscapes provide different amounts and patterns of ES under different future alternative scenarios (Goldstein et al., 2012; Lawler et al., 2014).

Several spatially-based decision support tools have emerged for ES assessment (Bagstad et al., 2013). Freely available tools such as InVEST (Integrated Valuation of Ecosystem Services and Tradeoffs), ARIES (ARtificial Intelligence for Ecosystem Services), EVT (Ecosystem Valuation Toolkit), TESSA (Toolkit for Ecosystem Service Site-based Assessment), and SolVES (Social Values for Ecosystem Services) have been developed and tested in private and public environmental decision contexts (Bagstad et al., 2014; Peh et al., 2013; Ruckelshaus et al., 2013; Sherrouse et al., 2011; Villa et al., 2014). But there has not yet been a comprehensive, systematic appraisal of the use of these tools. In order to improve user support and expand the reach of ES tools, it is vital to track how, where, and when they are being used.

This study examines the emerging user network of one particular tool – InVEST. We analyze where users are, which models they run, and country-level factors associated with model usage over a 25-month period. InVEST, developed by the Natural Capital Project, provides a suite of software models that can be used to map ES values and compare trade-offs among development alternatives (Kareiva et al., 2011; Nelson et al., 2009).

Research has found global patterns in where certain kinds of conservation activities and needs occur. Countries in which conservation activities are more likely to happen (based on lower protected area management costs, high numbers of endangered species, and identification as important for conservation) also score poorly on measures of corruption (McCreless et al., 2013; Smith et al., 2003, 2007). Other studies have found

that national characteristics such as number of threatened species, quality of governance, and deforestation rates are associated with the location of REDD demonstration sites and forest carbon projects (Cerbu et al., 2011; Lin et al., 2012; Lin et al., 2014). These studies suggest that country-level factors could be associated with conservation science uptake.

Building from this theory, we hypothesize that countries with high use of ecosystem service models also tend to have more capacity, more effective governance, lower environmental quality (Cerbu et al., 2011; Lin et al., 2012; Lin et al., 2014), more conservation spending, and more civic involvement in conservation (Table 1). We also make two more specific hypotheses: first, that biodiversity-related InVEST model usage is associated with lower environmental quality; and second, that carbon-related model usage occurs in countries with more forests and more overall conservation spending. Finally, we explore in more detail the effect that formal trainings have on use of ES models. We hypothesize that the average use in countries with trainings is higher than in countries without trainings, and that usage increases for a prolonged time period following trainings.

Support of these hypotheses would indicate certain conditions that facilitate the adoption and use of science-based tools. This understanding can help to predict patterns of uptake for new tools and target capacity building efforts to increase scientific and policy impact.

4.3. Methods

4.3.1. InVEST Data

When an InVEST model is run on a computer connected to the Internet, a log is created with date, IP address, InVEST version, and model type. We analyzed 25 months of these logs from June 2012 through June 2014. These data represent a network of InVEST usage, however they do not include models runs for computers not connected to the Internet, model activity done from outside the user interface (for example, through Python scripts), or certain specific models that were not reporting usage information during the timeframe of this study (such as coastal protection). We estimate that our dataset of 43,363 model runs represents most InVEST model runs during the study period.

We used IP addresses and GeoIP2 Precision Services provided by MaxMind to identify the country in which each model run occurred. We used model type to identify which type of ES model was run and we collapsed all possible model types into a concise list of primary ES models (Supporting Information).

We used information about the InVEST version of each model run to exclude model development and testing activity. For part of the analysis, we also screened out use that occurred in the U.S. because a) much of this use was likely internal Natural Capital Project scientists, and b) we did not want our results to be skewed by the fact that the bulk of use was in the U.S. We focused on terrestrial/freshwater models rather than marine because many of the marine models were not tracked over the entire study period and the available country-level environmental quality data are about forests and terrestrial biodiversity. The use of marine models at trainings in Portugal, Korea, Mexico, and Canada was not included in our data, so these are conservative estimates of the ES model use that occurred in those places.

4.3.2. Country-level Data

We gathered 19 country-level variables from global datasets (roman numerals below) and grouped them into 5 categories that we hypothesize are associated with use of ES models.

Capacity

We used estimates of i) population and ii) GDP/capita (in current US dollars) for the year 2013 available online from World Bank Open Data (<http://data.worldbank.org/>). iii) We used 2013 estimates of the number of Internet users per 100 people from World Bank Indicators available online (<http://data.worldbank.org/indicator/IT.NET.USER.P2>). iv) We used the percentage of households with a computer for the most recent year within the 2008-12 range for which data are available. These data were collected from national statistical offices by the International Telecommunication Union, the UN specialized agency for information and communication technologies. More information about the Internet and Computer Technology Data and Statistics Division is available online (<http://www.itu.int/en/ITU-D/Statistics>). v) We included a variable to indicate whether a country had a training prior to or during the study period. This variable was 0, 1, 2, or 3 based on the length of the training (0 if there was no training, 3 if there was a training for 3 or more days).

Governance

The World Bank estimates Worldwide Governance Indicators at the country level through surveys and consultations with citizens, experts, businesses, and international organizations (Kaufmann et al., 2011). We used three governance indicators relevant to the management of ESs: vi) government effectiveness (the quality of policy formulation

and implementation, vii) regulatory quality (the ability of governments to create policies and regulations to promote private industry), and viii) control of corruption. We used 2013 estimates of percentile (0-100%) rank among all countries for these two indicators. Data and background information for the indicators are available online (www.govindicators.org).

Environmental Quality

The general level of environmental quality for a country is difficult to measure and quantify. We focused on three main datasets for quantifiable, comparable information about environmental quality for countries. Using information from multiple sources allowed us to minimize the bias associated with any one dataset.

ix): The Global Environment Facility Benefits Index for Biodiversity is “a composite index of relative biodiversity potential for each country based on the species represented in each country, their threat status, and the diversity of habitat types in each country. The index has been normalized so that values run from 0 (no biodiversity potential) to 100 (maximum biodiversity potential)” (Pandey et al., 2006). Data and background information are available online from World Bank Open Data.

x-xii): The Environmental Performance Index (EPI), estimated by the Yale Center for Environmental Law and Policy, ranks how well countries protect ecosystems and protect human health from environmental harm (Hsu et al., 2013). We used x) overall EPI estimates for 2014, as well as two sub-indicators related to xi) forests (percent change in forest cover between 2000 and 2012 in areas with greater than 50% tree cover) and xii) biodiversity and habitat (an averaged composite of indices for critical habitat protection, terrestrial protected areas with national biome weight, terrestrial protected

areas with global biome weight, and marine protected areas). Data and background information for the indicators are available online (<http://epi.yale.edu/>).

xiii): Threatened mammal species includes the number of mammal species (excluding whales and porpoises) classified by the International Union for Conservation of Nature (IUCN) as endangered, vulnerable, rare, indeterminate, out of danger, or insufficiently known. We used 2014 estimates for each country provided by the United Nations Environmental Program, the World Conservation Monitoring Center, and the IUCN Red List of Threatened Species. Data and background information are available online from World Bank Open Data.

Conservation Spending

Two datasets provided information on country-level spending on conservation:

xiv) We used information about total average annual spending (in \$ US million 2005) from 2001-2008 as estimated by Waldron et al. (2013). This includes all flows of funding estimated: international donors, domestic governments, trust funds, and self-funding via user payments. We also used a database of REDD+ projects sourced through a number of dedicated multilateral and bilateral climate funds to include xv) the amount of climate finance and xvi) the amount of REDD funding countries received from 2003-2013. Data and background information are available online (<http://www.climatefundsupdate.org/data>).

Civil Society Involvement in Conservation

We followed McCreless et al. (2013) and focused on three datasets that measure the extent to which civil society is involved in conservation efforts for many countries.

xvii) BirdLife International (BLI) is the largest global partnership of conservation

organizations in the world. We used data on citizen membership in BLI partner organizations available online (www.birdlife.org/worldwide/national/index.html). We standardized NGO membership numbers by country population to represent the proportion of a country's population that belongs to a leading local conservation NGO.

xviii) IUCN is the largest global environmental organization in the world. The Environmental Sustainability Index provides a country level estimate of the number of IUCN organizations per million people. These data are available online (www.yale.edu/esi/c_variableprofiles.pdf).

xix) Local Agenda 21 initiatives are “measures undertaken and overseen by local authorities to address problems of environmental sustainability, and represent the involvement of civil society in environmental governance.” The Environmental Sustainability Index provides a country level estimate of the number of local Agenda 21 initiatives per million people available online (www.yale.edu/esi/c_variableprofiles.pdf).

4.3.3. Analysis

We examined the relationships that use of ES models has with each of the country-level variables. We also examined relationships that carbon model use and habitat model use have with particular variables. We used nonparametric Spearman rank correlations because the individual datasets did not meet the assumptions required for parametric correlations such as data having normal distributions (Crawley, 2007). All data analyses were conducted in the statistical platform R (R, 2011).

We tested for correlation among country-level variables and used principal components analysis (PCA) to reduce two groups of highly correlated variables with

correlation coefficients > 0.70 : Governance and Biodiversity (Dormann et al., 2013). We used the first principal component to capture over 90% of the variance for two groups of highly correlated variables (Table 2). We then used model selection to rank all possible linear combinations of variables to identify those statistical models that could best explain the outcome variable of InVEST model usage. We used the R package MuMIn (multi-model inference) for model selection, and ranked models based on AICc values to identify the explanatory variables present among the top models (Barton, 2014). We did model selection with only the variables that have data for all countries so that submodels would not be fitted for different datasets. All possible combinations of our 8 predictor variables resulted in 256 models being evaluated.

For our analysis on trainings, we initially narrowed our focus to 9 trainings that occurred within our study period and that had at least 10 model runs within 30 days before and after the training (Table 4). We defined InVEST trainings as organized meetings of Natural Capital Project staff with registered event participants in order to introduce and train people in the use of InVEST models. We gathered information on trainings from Natural Capital Project records, including meeting agendas, participant lists, training evaluation surveys, and facilitator notes.

We defined a “Before” period of time as the 13 weeks (approximately 90 days) before a training, “After 1” as the 13-week period following a training, and “After 2” as the next 13-week period (Figure 4). We computed a change factor as the ratio of model runs between the “Before” and “After 1” periods. We also calculated average weekly usage in these time periods to estimate the prolonged effect of trainings on InVEST model usage.

To evaluate the effect of trainings statistically, we created a generalized linear mixed model with country as a random effect and period as a fixed effect. We assumed a Poisson distribution for weekly usage data and tested whether usage was different in the periods “Before”, “After 1”, and “After 2” with a type III Wald chisquare test.

Finally, we combined our quantitative results with a qualitative analysis of documents from these trainings, including detailed facilitator notes and participant surveys (Creswell, 2009). This review focused on lessons learned by the training facilitators and the following open-ended questions given to participants:

- Please identify two things you found the most useful from this course (favorite parts).
- What recommendations do you have for improving the course (least favorite parts)?
- What subject(s) would you like to see offered in future training sessions?
- Do you have any additional comments?

4.4. Results

The use of InVEST models increased over the 25-month study period (Figure 1). A significant amount of overall use occurred in the U.S., but most of the growth over time occurred in non-U.S. countries.

Across all countries, ES models related to habitat, water yield, carbon, sediment, and nutrients were used most often (Figure 2). Based on the ES classifications provided by the Millennium Ecosystem Assessment (Program) (2005), we identified each service as provisioning, regulating, supporting, or cultural and found that 46% of model use was for regulating services (see SI for how InVEST models represent each service type).

We found a number of positive and significant correlations between total use of ES models and the country-level variables (Table 2). These relationships exist in all of the hypothesized categories, especially for variables related to Capacity. For carbon model use, none of the hypothesized relationships showed positive correlations. For habitat model use, the Biodiversity PC and EPI variables had positive and significant correlations. We examined whether trainings were more likely to occur in particular kinds of places and did not find that trainings were significantly correlated with any of the other country-level variables.

Model selection shows which variables are present in the statistical models that best explain InVEST usage (Table 3). “Population,” “Training,” and “Internet” were found to be the most common and important variables in the top 10 models. Other variables such as those for biodiversity and the Environmental Performance Index appear in some statistical models, but not with the same consistency.

In analyzing the effect of trainings, we found that the average use of ES models in countries with trainings was over two times larger than in countries without trainings (Figure 3). For the 9 trainings we analyzed, we typically observed a burst of usage during the training and then more activity in the After 1 periods than the Before periods (Figure 4 and Supporting Information). The change factors (ratio of total model runs in After 1 period to model runs in Before period) showed that all but one of the cases had an increase in model use following a training (Table 4). The average change factor for 1-day trainings was 0.80 (n=2), 2-day trainings was 6.2 (n=3), and 3-or-more-day trainings was 3.5 (n=4). Using the generalized linear mixed model with country as a random effect, we

found a positive and significant effect of trainings on model use (Figure 5. $\chi^2_2 = 154.8$; $p < 0.001$).

The qualitative review of training evaluations further illuminated place-specific factors that help explain use of ES models, such as connections to a university course or a funded project with deadlines. Among the 7 cases reviewed, change factors were highest for trainings that offered case studies and demonstrations relevant to participants' on-going projects and when a specific deliverable using InVEST was due soon after the event. Training attendees valued the opportunity to interact with experienced analysts and developers of the InVEST models. This in-person support served to narrow the user-developer divide that is a known barrier to decision-support tool uptake (Haklay & Tobon, 2003). After establishing a rapport with training facilitators, participants felt more comfortable applying the tool and requesting support following the event.

4.5. Discussion and Conclusion

Using a unique global dataset, we conduct the first quantitative analysis of the use of ecosystem service modeling tools. We show that general capacity to use these kinds of tools (i.e., population, GDP/capita, computer technology) as well as specific capacity building for the tool in question (i.e., trainings) are the strongest predictors of model use in a given country. While the effect of trainings varied widely among countries, in general trainings had a positive and enduring effect on tool use. Understanding the factors that encourage uptake and use of scientific tools can help target trainings, improve tool design, and improve impact of scientific knowledge on decisions.

Formal trainings can build on a country's existing capacity to use decision support tools. Why do some trainings have a bigger impact than others? Trainings that are longer than 1-day probably have a larger effect because they allow trainers and participants to spend more time learning the details of how to use the tools. Extended trainings include time for repeatedly running models and practice working through the different steps of an ES assessment (Rosenthal et al., 2014). Other, site-specific factors related to a training can explain the places where we observe larger impact, such as countries with reliable Internet access and more spending on conservation-related research. Regardless of these local factors, problem-based exercises that are simple, well designed and include detailed guidance (e.g., step-by-step tutorials) show promise as an entry point for a range of potential tool users (Verutes & Rosenthal, 2014). Further research on effective ways to sequence introductory to more technical content for a diverse range of audiences can inform creative approaches to building local capacity that actively engage participants in learning a new technology.

Can country-level variables help us identify new, underserved audiences for computer-based decision support tools, or understand which technologies may be most relevant to a local place? We found evidence that country-level conditions can be used to estimate the capacity for using InVEST models generally, but did not find that country-level conditions were highly correlated with the use of carbon or biodiversity ES models. It is likely that other variables beyond those we tested are associated with tool use. These include some of the site-specific factors uncovered in our qualitative assessment, such as a funded ES project with deadlines, as well as factors related to the presence of ES

concepts in government policies, overall amounts of science research activity, and levels of education in a population.

Limitations to this study are worth noting. First, as mentioned above, some ES models (i.e. marine models) were not included in our data set, so we almost certainly underestimated the impacts of trainings that focused in part on these models. Second, our data capture where a given model is used, not the location where ES are being evaluated. Researchers in one country can run InVEST models focused on another, and some InVEST trainings included participants from other countries, who likely went on to use InVEST outside the country where the training was held. Excluding the US from our analyses removes the largest source of this issue, but further subsetting to countries where we are certain models are being used in-country reduces the size of our dataset rapidly. Our analyses therefore pertain to where models are used rather than where they are applied. Third, our analyses focused on InVEST, but of course several other ES models and computer-based tools exist. We are not aware of equivalent tracking data for any other tool, but we expect they would display patterns associated with national-scale capacity and training opportunities.

Future research in this area would benefit from in-depth qualitative analyses to better understand the factors that lead to differences in effectiveness of trainings. In addition, understanding the relationships among users (e.g., through surveys of users) could help to illuminate how technology diffuses through social networks (Haythornthwaite, 1996). User surveys could also help to clarify user demographics, the decision contexts in which the tools are used, and the ways in which outputs are used to inform decisions (McKenzie et al., 2014). Finally, model developers could make several

simple additions to the information reported for each model use, including location of the region being assessed, and saving records for later reporting if the computer is not connected to the Internet.

People may use an ES model for any of a number of reasons: they receive training, they find decision support tools useful for a particular context, they have project deadlines or an academic adviser nudging them to produce results, etc. Efforts to increase use of these models should therefore focus equally on understanding these drivers, building capacity generally, and providing specific training in the tools. Formal training opportunities that provide locally-relevant demonstrations of the tool and follow-up activities to reinforce what was learned are an effective way to support the continued usage of spatial models in ES assessments. If country-level factors can predict use along with trainings, then we need to be aware of which countries have the basic capacity to use these models. And we need to think about more than just trainings – having key conditions in place, such as the capacity variables illuminated in this study, will enable people to use what they learn. Tracking and explaining tool use can lead to more strategic deployment of technology and smarter applications of these models for informing real world decisions..

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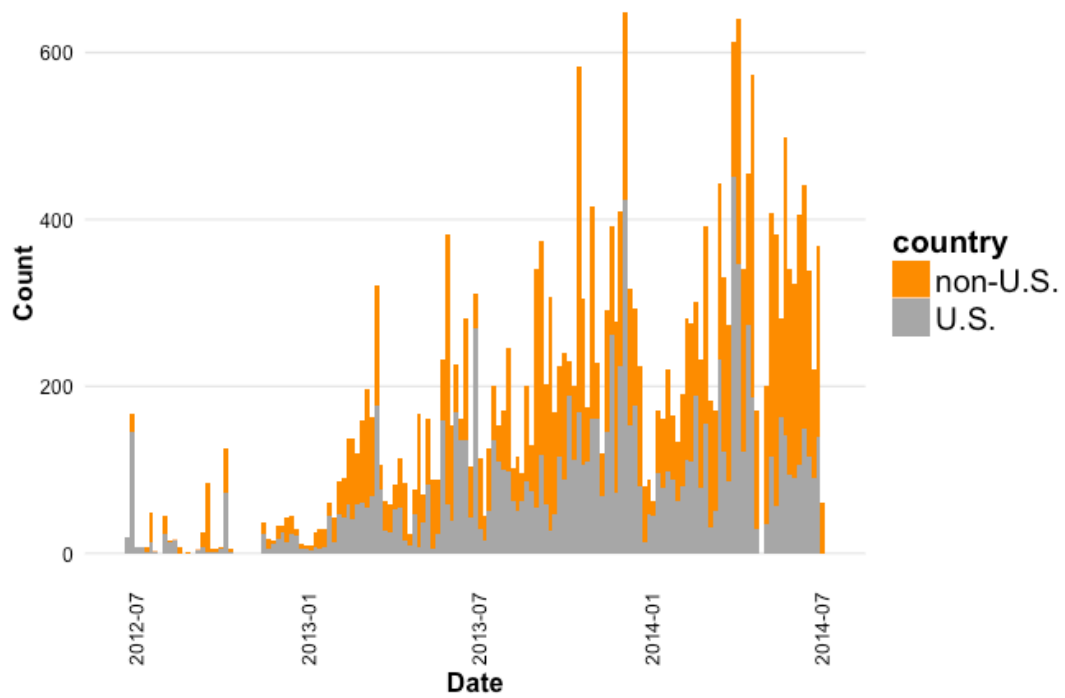


Figure 4.1: Growth in the use of InVEST models over time. Model use occurred in 102 different countries with 44% of all use occurring in the U.S. For non-U.S. countries, there were 14,301 model runs with 43% of use occurring in 5 countries: the UK (1554), Germany (1491), China (1209), France (1074), and Colombia (780).

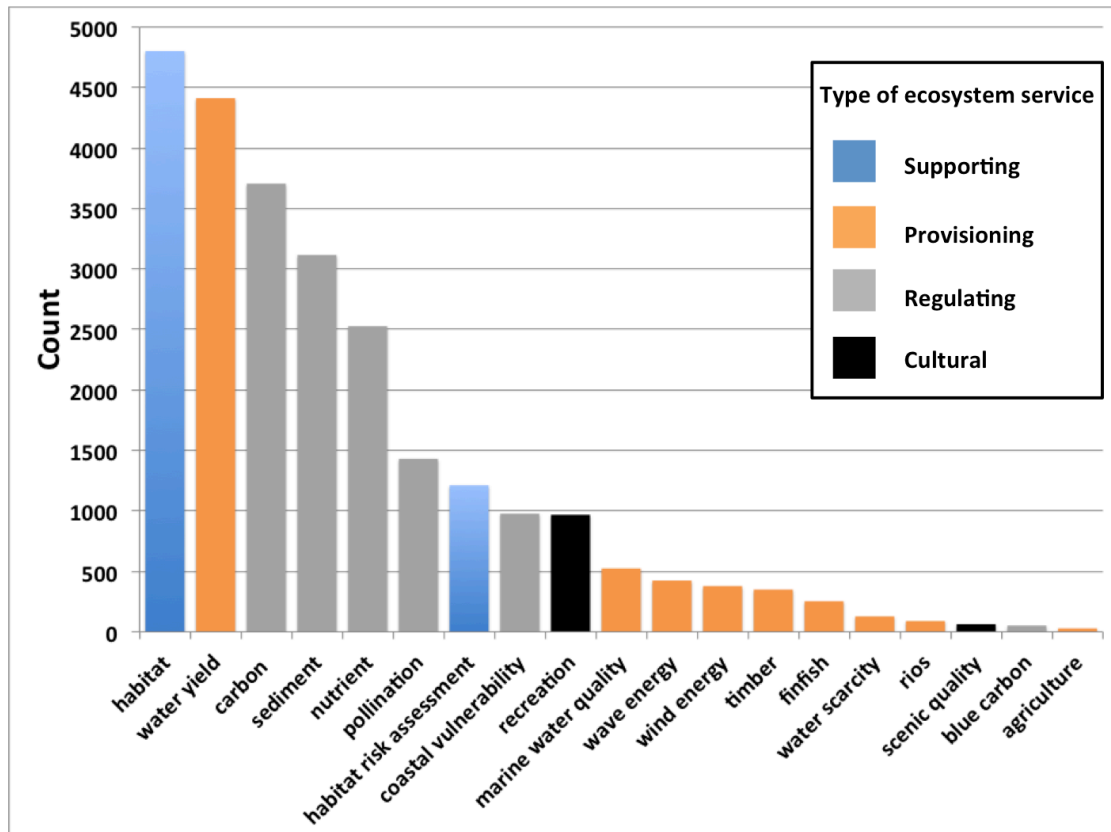


Figure 4.2: Types of ecosystem service models used. 46% of all model use is for regulating services. “Rios” focuses on freshwater provisioning services but includes other service types as well.

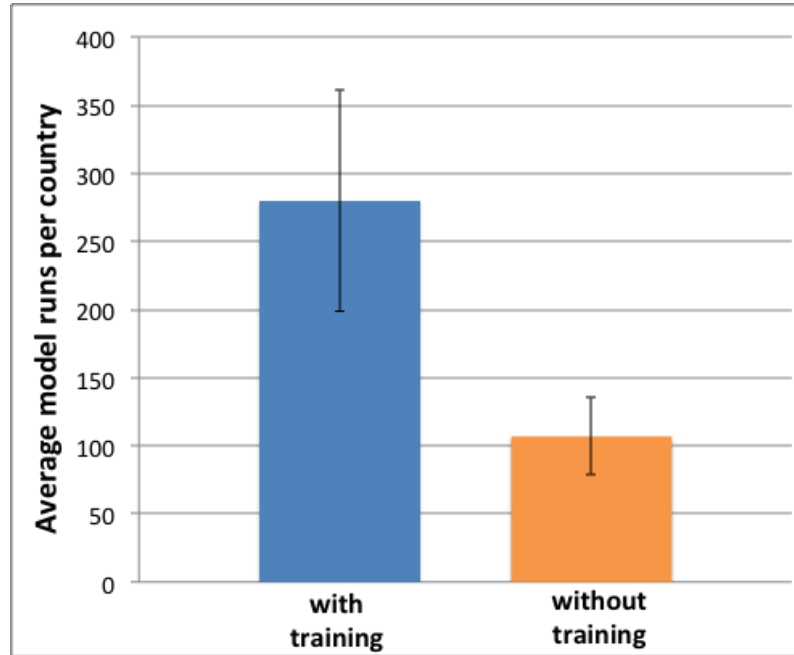


Figure 4.3: Comparison of average use in non-U.S. countries with and without trainings. Error bars depict standard error. There was a significant difference in the average use for countries with (mean = 280, sd = 81) and countries without (mean = 107, sd = 29) trainings ($t(22.6) = 2.0$, $p = 0.05$).

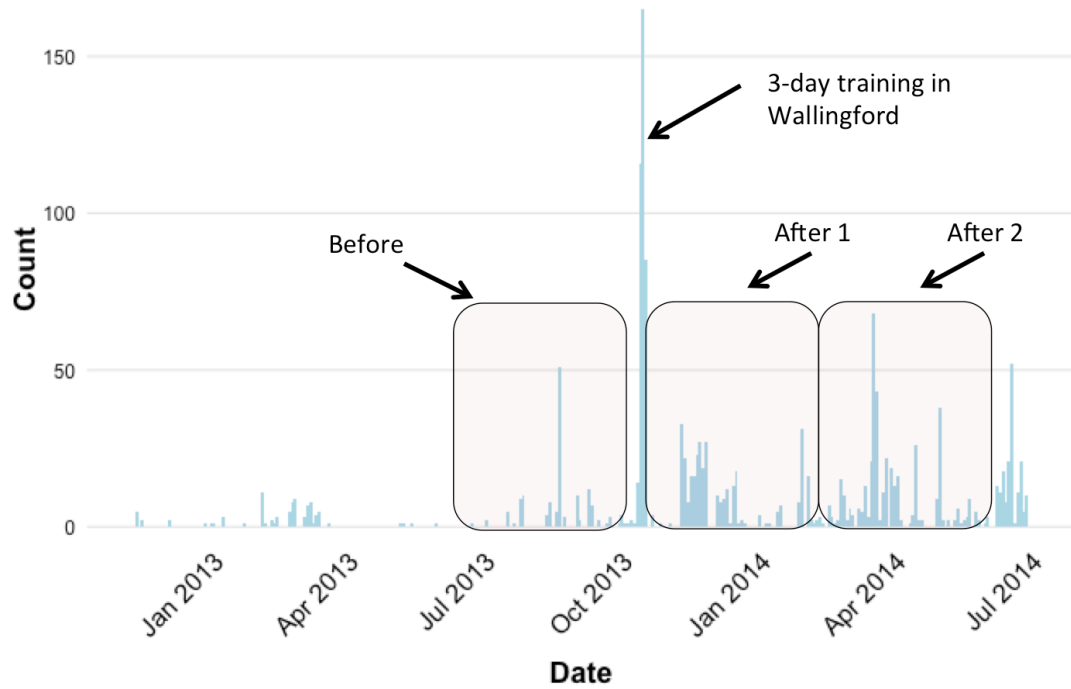


Figure 4.4: Example of a training in the UK. Model use spikes during the training. We compared the number of models runs and average weekly use for one 13-week period before and two 13-week periods after trainings.

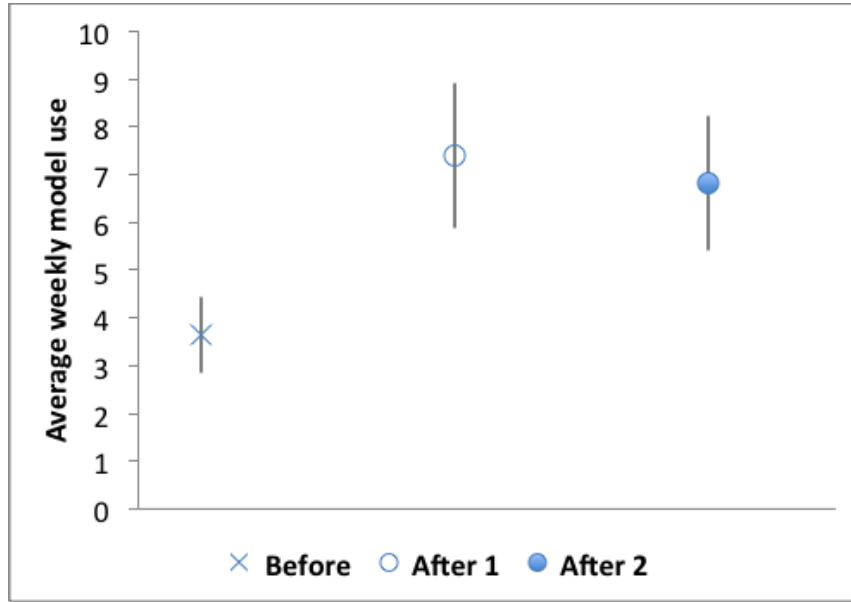


Figure 4.5: The positive effect of trainings on model usage across 9 cases. “Before” is average weekly model use and standard error for a 13-week period (approximately 90 days) before a training. “After 1” is for a 13-week period after a training and “After 2” is for the following 13-week period.

Table 4.1: The main hypothesized drivers of ecosystem service model use, predicted relationships, and justifications.

Category	Relationship with use of ES models	Justification
Capacity	+	Places with more people, trainings, and access to technology have more basic capacity to use ES models
Governance	+	Stronger systems of governance enables more use of sophisticated decision support tools
Environmental quality	–	Worse environmental quality makes it more likely that people will use tools to inform environmental decisions
Conservation spending	+	Places with higher levels of conservation spending have an established presence of environmental organizations and a higher likelihood that ES models will be used
Civic engagement in conservation	+	People in places with higher rates of involvement in conservation organizations are more likely to use ES models

Table 4.2: Correlation results comparing model use with country level variables. Bold rows indicate positive and significant correlations. N is the number of countries for which those data are available. WE calculated the first principal component of Governance and Biodiversity variables to capture >90% of the variance in the underlying variables. “Governance PC” includes government effectiveness, regulatory quality, and control of corruption (for pairwise correlations, rho=0.91, 0.95, and 0.87); “Biodiversity PC” includes GEF Benefits Index of Biodiversity and mammals (rho=0.77).

**** p < 0.01; * p < 0.05; + p < 0.1**

Variable comparison	N	Spearman’s rho	95% CI	p-value
<i>Total model use ~ ...</i>				
CAPACITY				
GDP per capita	94	0.347	(0.155, 0.513)	0.00062 **
Population	94	0.482	(0.309, 0.624)	1.8E-7 **
Internet	94	0.316	(0.121, 0.487)	0.0019 **
Computers	86	0.237	(0.0269, 0.428)	0.028 *
Trainings	94	0.273	(0.0748, 0.451)	0.0077 **
GOVERNANCE				
Governance PC	94	0.335	(0.142, 0.504)	0.00095 **
ENVIRONMENTAL QUALITY				
Biodiversity PC	94	0.317	(0.122, 0.488)	0.0019 **
EPI	94	0.262	(0.062, 0.441)	0.011 *
EPI Forests	86	-0.0179	(-0.229, 0.195)	0.87
EPI biodiversity	94	0.131	(-0.0732, 0.325)	0.21
CONSERVATION SPENDING				
Conservation spending	92	0.603	(0.454, 0.719)	2.1E-10 **
REDD	31	0.0580	(-0.136, 0.540)	0.76
Climate finance	94	0.00992	(-0.193, 0.212)	0.92
CIVIC ENGAGEMENT IN CONSERVATION				
BLI	62	-0.248	(-0.468, 0.00241)	0.052 +
IUCN	89	0.0885	(-0.122, 0.291)	0.41
Agenda 21	74	0.0867	(0.000496, 0.435)	0.46
<i>Carbon model use ~ ...</i>				
EPI Forests	86	0.0252	(-0.188, 0.236)	0.82
REDD	31	0.127	(-0.238, 0.461)	0.50
Climate finance	94	0.0311	(-0.173, 0.232)	0.77
<i>Habitat model use ~ ...</i>				
Biodiversity PC	94	0.244	(0.043, 0.425)	0.018 **
EPI	94	0.236	(0.0352, 0.419)	0.022 *
EPI Forests	86	0.0153	(-0.197, 0.226)	0.89
EPI biodiversity	94	0.153	(-0.0509, 0.345)	0.14

Table 4.3: Model selection results for the top 10 statistical models by AICc value. Statistical models are listed by row in rank order (the first has the lowest AICc value corresponding to best fit) with a check mark for variables that are included in the statistical model. Only the 8 variables that contained data for all countries were included in model selection.

Model	Biodiversity_PC	EPI	EPI biodiversity	GDP	Governance_PC	Internet	Population	Training	df	Log likelihood	ΔAIC_c
1	-	-	-	-	-	✓	✓	✓	5	-652.62	0.00
2	-	-	✓	-	-	✓	✓	✓	6	-652.25	1.55
3	✓	-	-	-	-	✓	✓	✓	6	-652.27	1.58
4	-	✓	-	-	-	-	✓	✓	5	-653.47	1.70
5	-	✓	-	-	-	✓	✓	✓	6	-652.52	2.09
6	-	-	-	-	✓	✓	✓	✓	6	-652.61	2.26
7	-	-	-	✓	-	✓	✓	✓	6	-652.62	2.27
8	-	-	-	-	-	✓	✓	-	4	-654.89	2.31
9	-	-	-	-	✓	-	✓	✓	5	-654.04	2.83
10	-	-	✓	-	-	✓	✓	-	5	-654.23	3.22

Table 4.4: Comparison of ES model use before and after trainings. Change factor is the ratio of use in 13-week periods after/before training.

Country	Date of training	Model use before training	Model use after training	Change factor
Argentina**	9/12/2013	14	52	3.71
Cambodia***	6/17/2013	6	13	2.17
Canada**	2/04/2013	64	92	1.44
Chile***	9/09/2013	17	82	4.82
Korea*	9/11/2012	46	54	1.17
Peru***	5/27/2013	18	40	2.22
Spain**	11/18/2013	13	175	13.46
UK1***	10/15/2013	137	661	4.82
UK2*	3/07/2013	50	23	0.42

* 1 day training, ** 2-day, *** 3-day or longer

4.8. Supporting Information

4.8.1. Ecosystem Service Models in InVEST

Table 4.5: Ecosystem service types for InVEST models.

InVEST model category	Ecosystem service type	Specific models included
Agriculture	Provisioning	"agriculture"
Blue carbon	Regulating	"blue_carbon" "blue_carbon_biophysical" "blue_carbon_preprocessor"
Carbon	Regulating	"carbon_biophysical" "carbon_biophysical\ "carbon_combined" "carbon_valuation"
Coastal vulnerability	Regulating	"coastal_vulnerability"
Finfish	Provisioning	"finfish_aquaculture"
Habitat	Supporting	"habitat_quality" "biodiversity_arc" "biodiversity_biophysical" "biodiversity_biophysical\ "biodiversity_valuation"
Habitat risk assessment	Supporting	"hra" "HRA_LaunchGUI_arc" "hra_preprocessor"
Marine water quality	Provisioning	"marine_water_quality_biophysical" "marine_water_quality_biophysical\ "marine_water_valuation"
Nutrient	Regulating	"nutrient"
Pollination	Regulating	"pollination" "pollination_biophysical" "pollination_valuation"
Recreation	Cultural	"recreation_client" "recreation_client_init" "recreation_client_scenario"
Rios	Provisioning	"rios" "rios_0.3.0" "rios_sediment"
Scenic quality	Cultural	"aesthetic_quality" "scenic_quality"
Sediment	Regulating	"sediment" "sediment_biophysical"
Timber	Provisioning	"timber" "timber\ "timber_valuation"
Water scarcity	Provisioning	"water_scarcity"
Water yield	Provisioning	"hydropower_valuation"

		"hydropower_water_yield" "water_yield" "water_yield\"
Wave energy	Provisioning	"wave_energy" "wave_energy_biophysical" "wave_energy_biophysical\ "wave_energy_valuation"
Wind energy	Provisioning	"wind_energy" "wind_energy_biophysical" "wind_energy_uri_handler"
<i>Removed from analysis, either with development activity or separately to filter out non-model use</i>		
Overlap	-	"overlap_analysis" "overlap_analysis_mz" "OverlapAnalysis_arc"
Scenarios	-	"scenario_generator"
Other model types in data log	-	"#VALUE!" "adept" "adept_core" "coastal_vulnerability_post_processing" "crop_production" "developme" "development" "fisheries" "GridSeascape_arc" "habitat_suitability" "malaria" "monthly_water_yield" "monthly_water_yield_old" "ntfp" "percent_land" "pollination_10_arc" "recreation_init" "recreation_scenario" "rios_beer" "rios_porter" "rios_rsat" "routedem" "scenario_generator_summary" "sdr" "test" "test_invest_2" "test_string_submission"

		"test!!! version number?" "viewshed_grass" "wind_energy_valuation"
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4.8.2. Model Use in Countries with Trainings

Model use over time and average weekly model use (with standard error) in 13-week periods before and after trainings. We analyzed the effect of trainings in detail for the following 9 trainings.

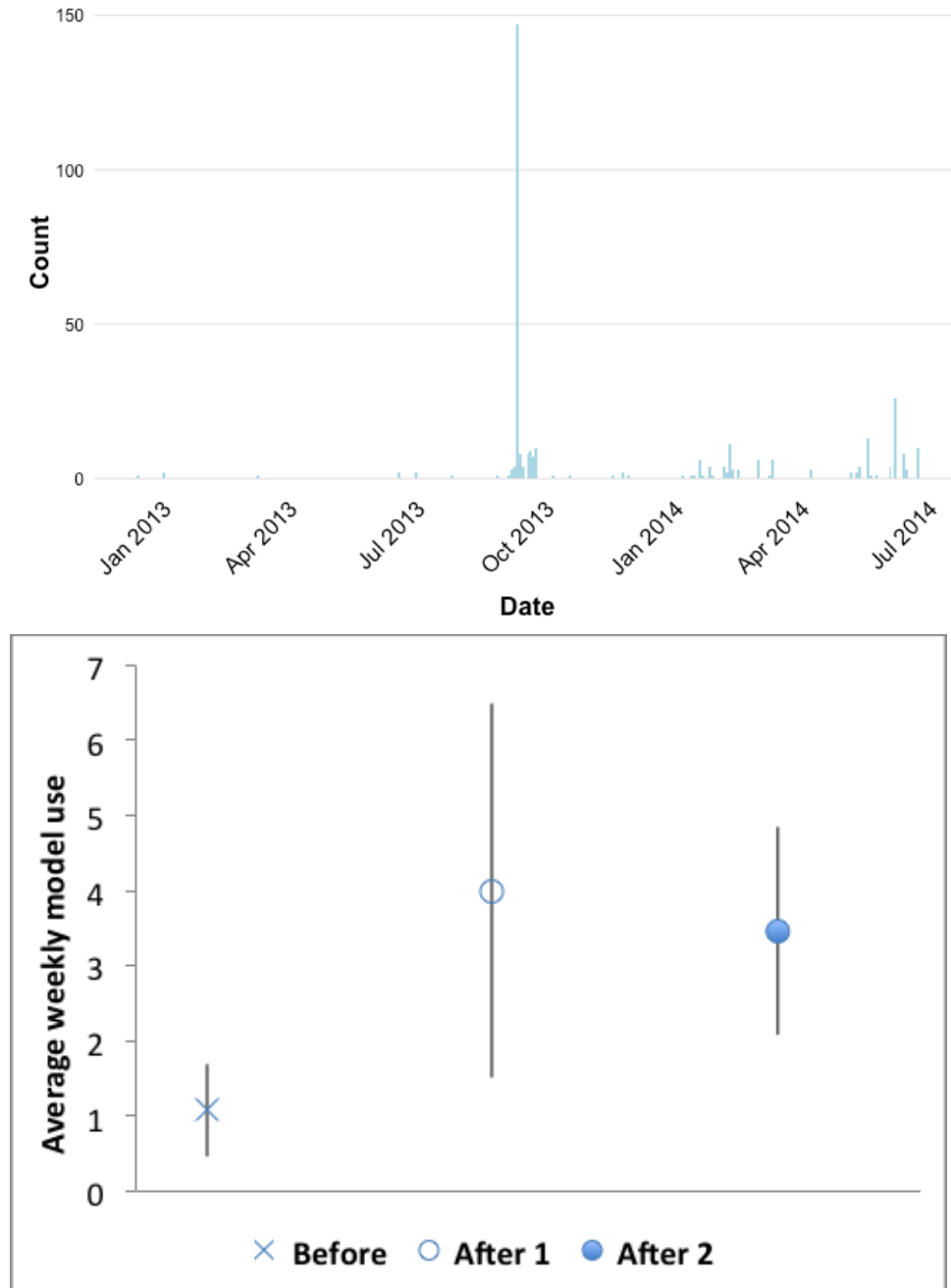


Figure 4.6: Argentina 2-day training 9/12/2013

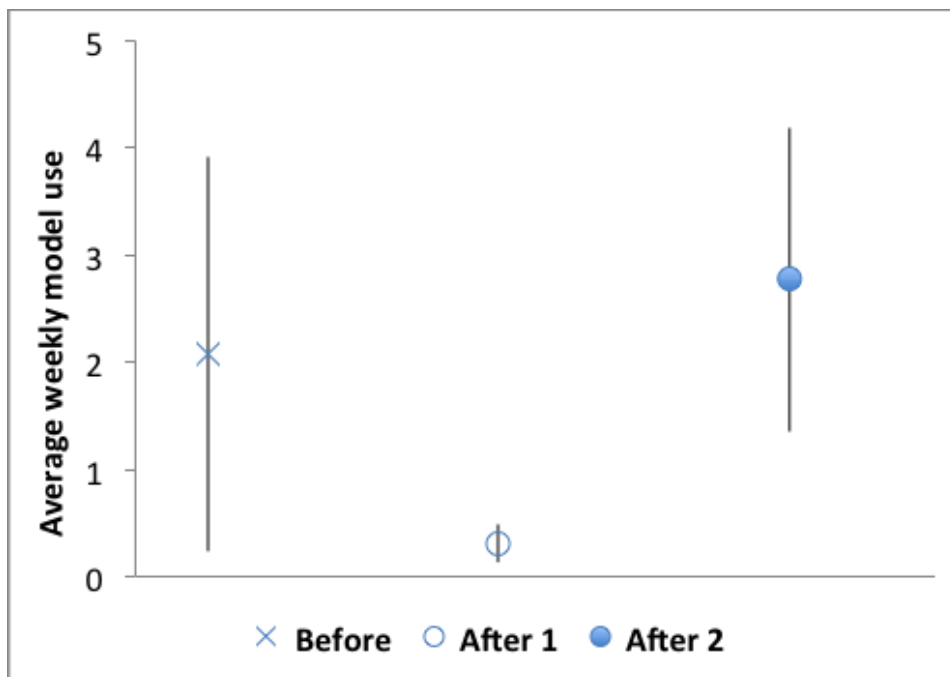
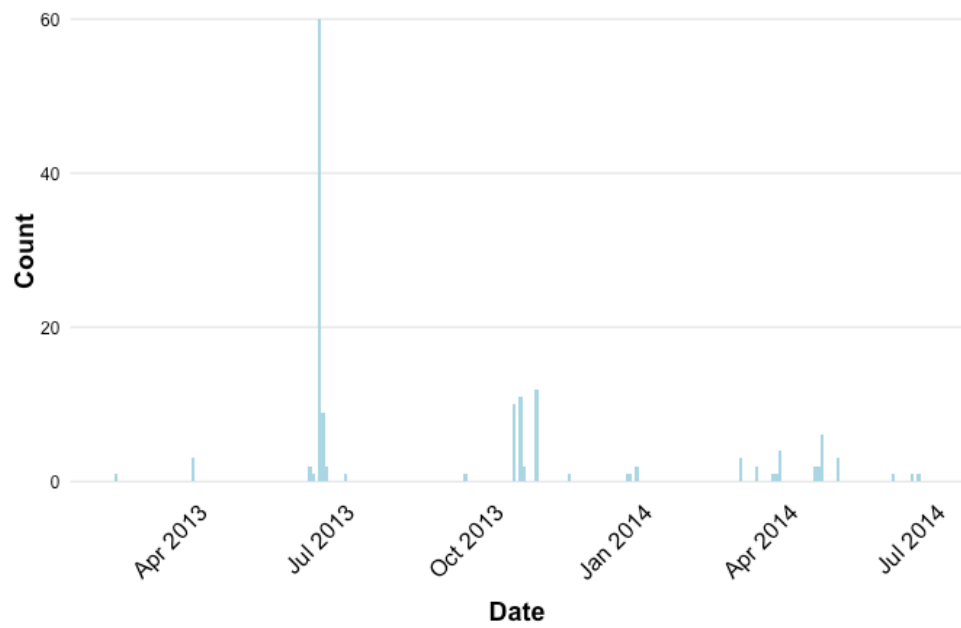


Figure 4.7: Cambodia 3-day training 6/17/2013

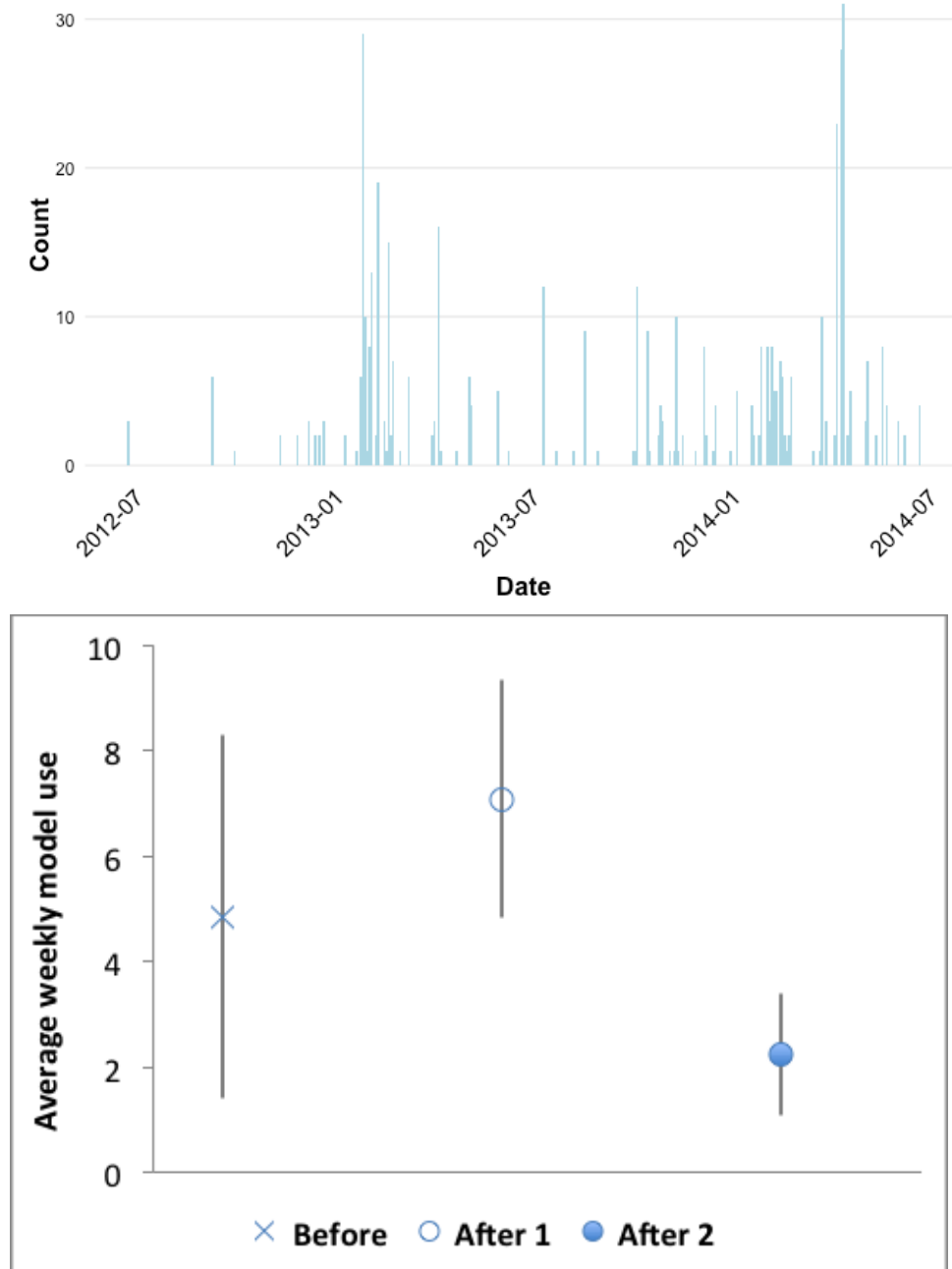


Figure 4.8: Canada 2-day training 2/04/2013

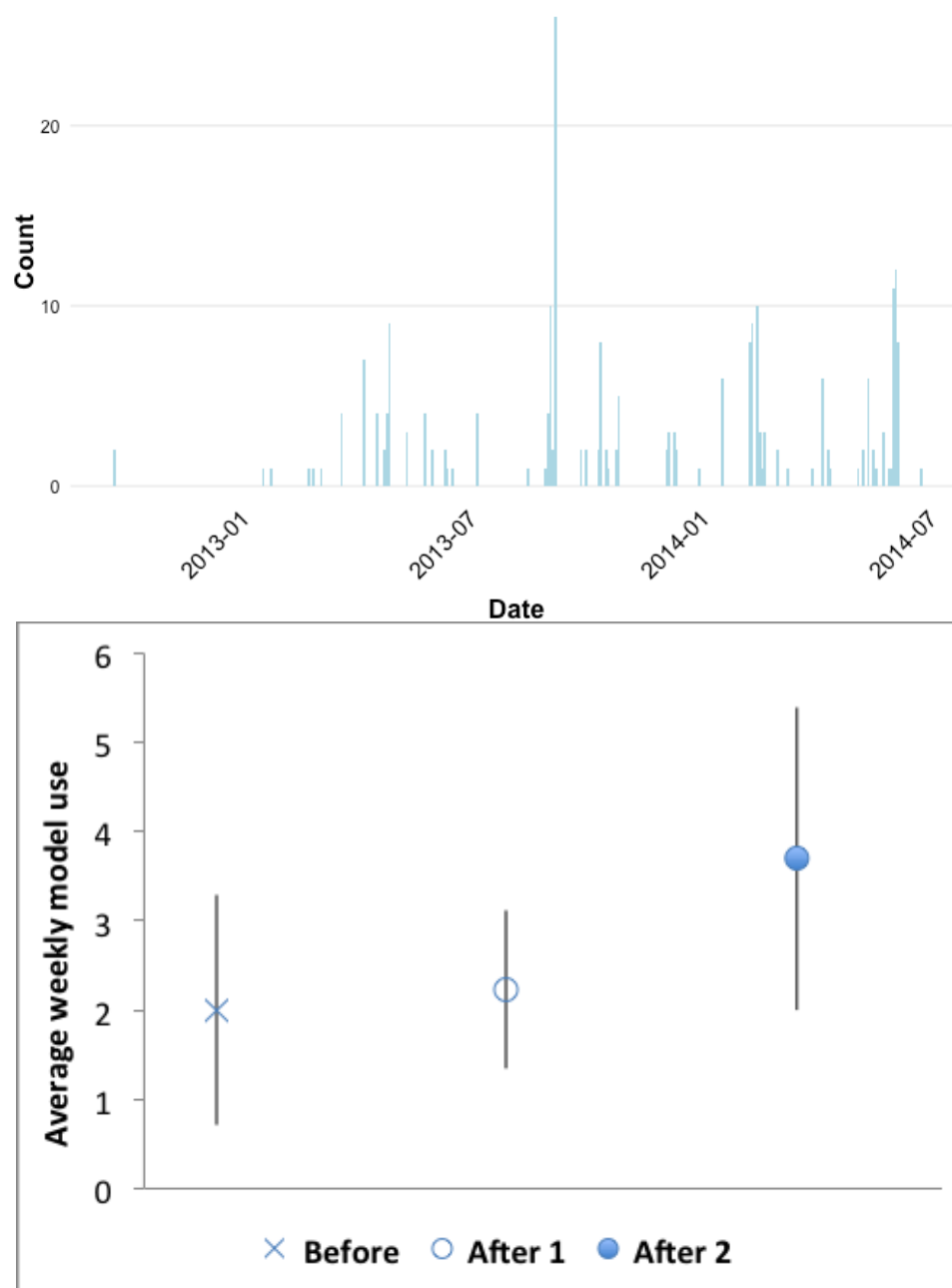


Figure 4.9: Chile 3-day training 9/09/2013

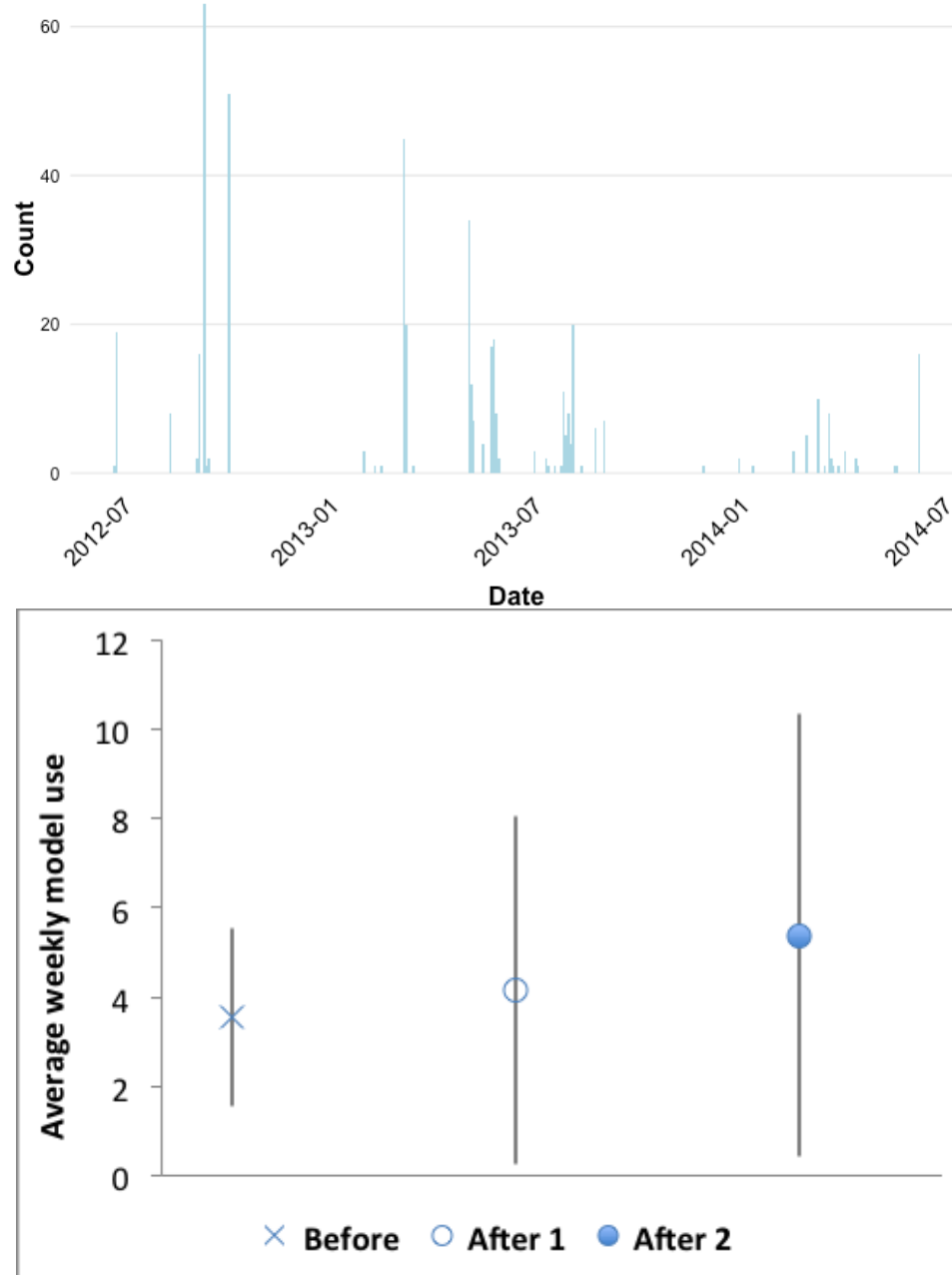


Figure 4.10: Korea 1-day training 9/11/2012

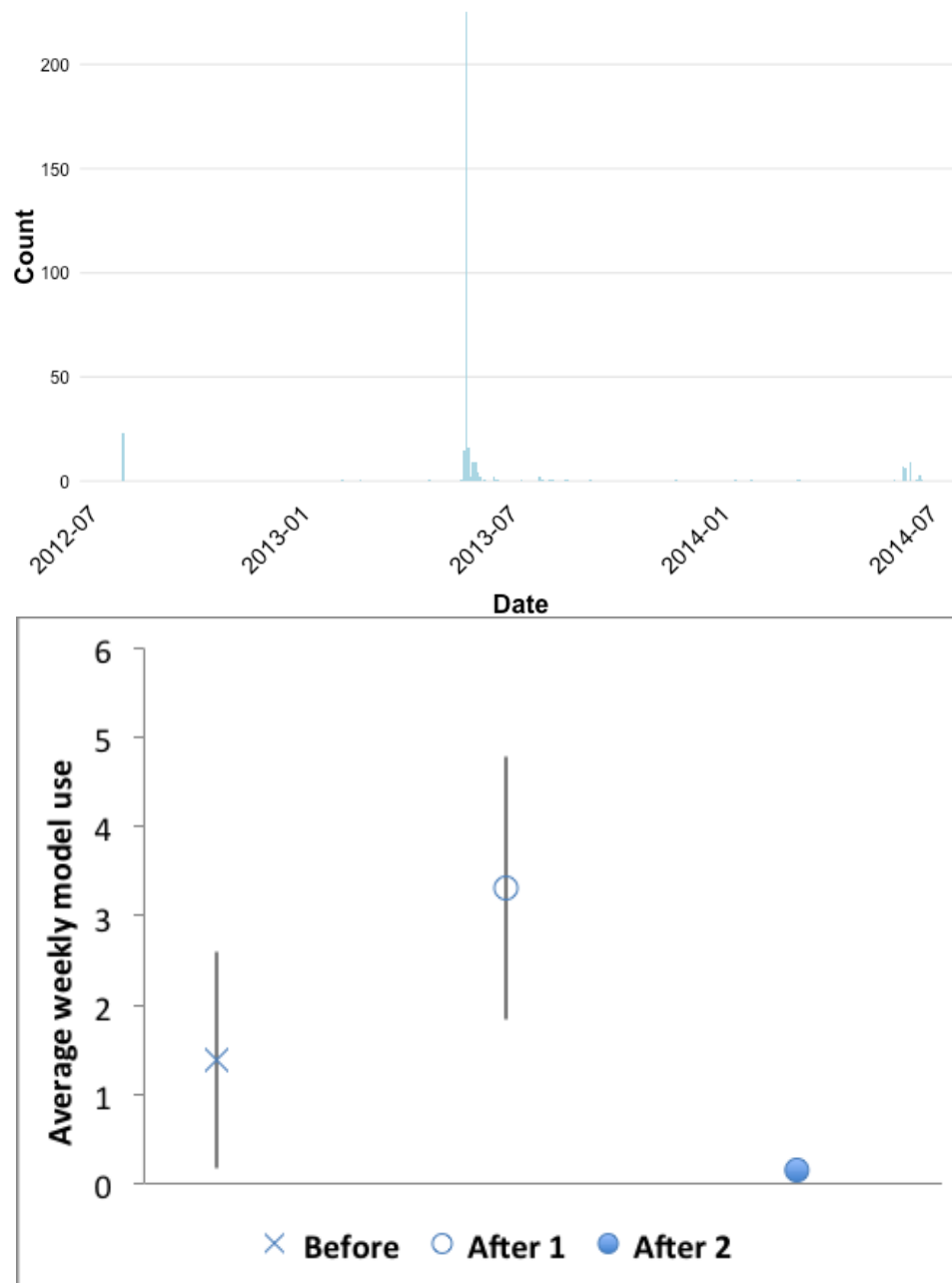


Figure 4.11: Peru 3-day training 5/27/2013

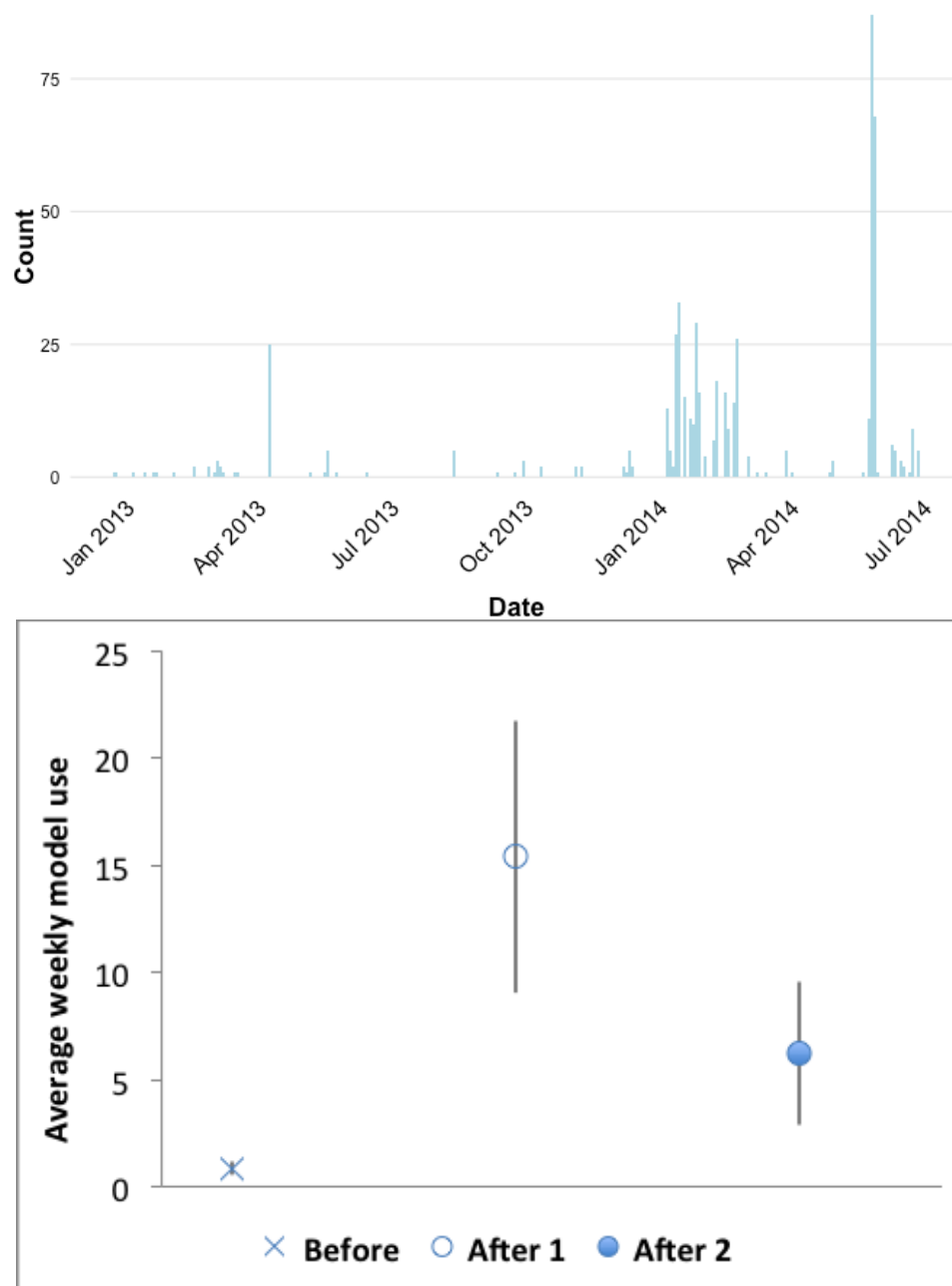
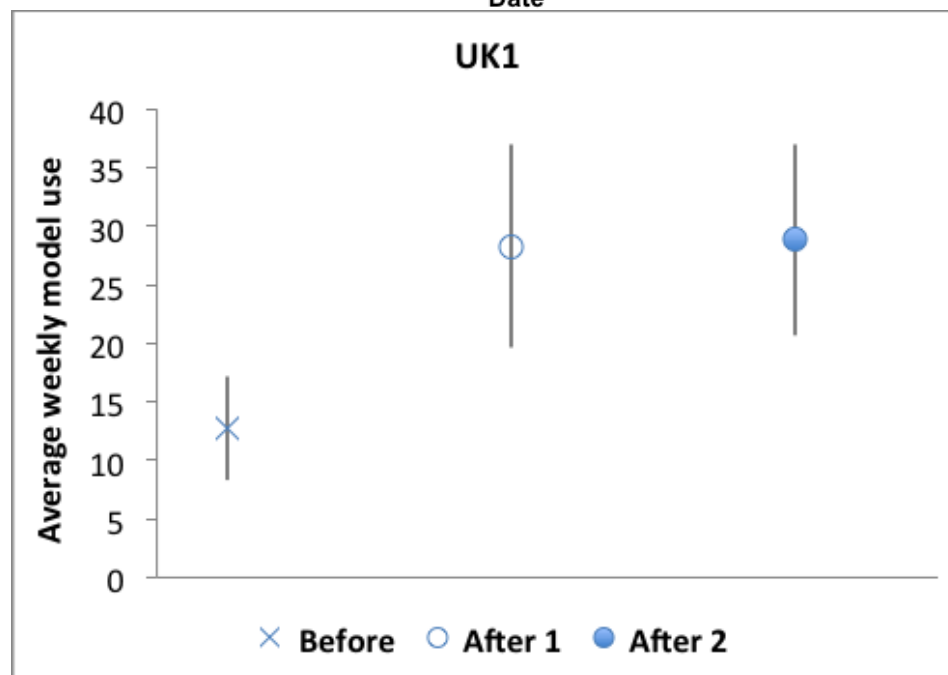
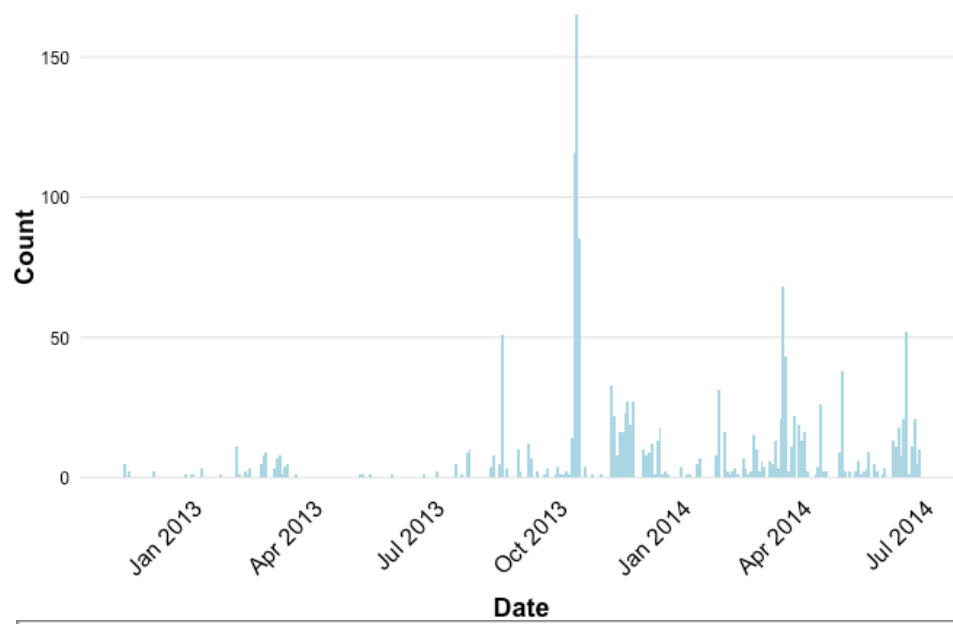


Figure 4.12: Spain 2-day training 11/18/2013



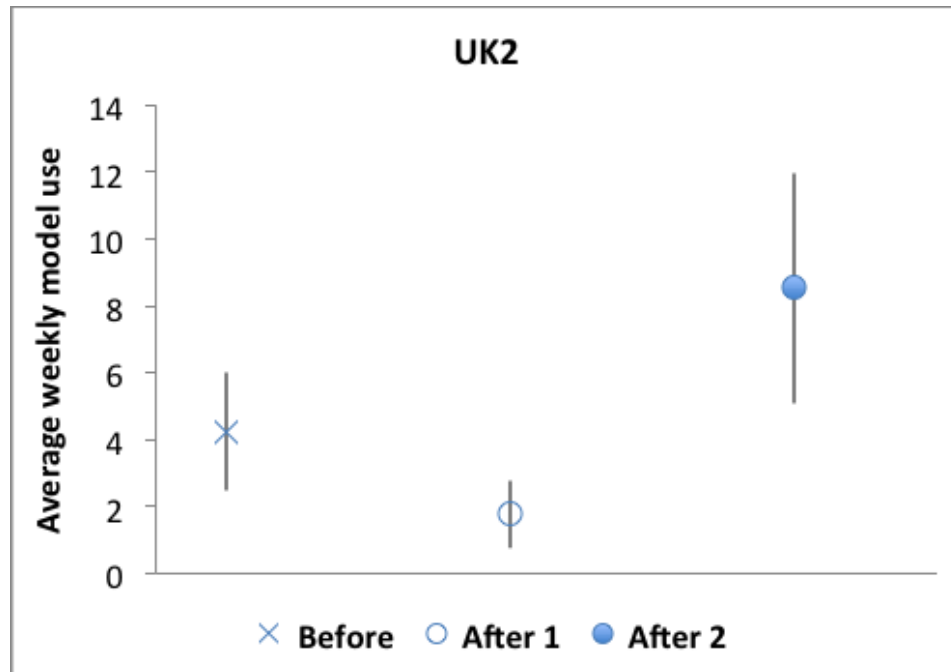


Figure 4.13: UK1 3-day training 10/15/2013 and UK2 1-day training 3/07/2013

CHAPTER 5: CONCLUSION

Many within the conservation community have embraced ecosystem services as a strategy for protecting and restoring nature. They aim to inform land use decisions about the value of nature. Two key assumptions underpin this strategy. The first is that incorporating information about the value of ecosystem services into decisions leads to improved environmental and human well-being outcomes. The second assumption, antecedent to the first, is that scientific knowledge about ecosystem services actually changes decisions. These assumptions imply that there is a set of different potential future states that society can choose from, and that some states are better or worse than others in terms of environmental quality and human well-being. The role of ecosystem services knowledge is to guide decisions that will result in a future state with better conditions.

In this dissertation, I focused on what difference ecosystem services knowledge actually makes to decision-making at regional and national levels. I have taken particular interest in how and why people responsible for making land use decisions change their minds and begin to consider new knowledge about the benefits provided by nature. I described only some of the many factors that make scientific knowledge about ecosystem services used and useful. I found signs that ecosystem services knowledge can significantly impact how people understand and frame environmental problems, evaluate options for how to respond to these problems, and decide which option or options to pursue. My work at the science policy interface fills the gap between scientists doing research on ecosystem services and the policy decisions they often hope to affect.

5.1. Legitimacy

In the chapter on “what explains the impact of ecosystem services knowledge on decisions,” I found legitimacy to be the most important explanatory variable of all the factors I analyzed. However, in the chapter on “evaluating the impact of an ecosystem service assessment,” legitimacy was not as prominent an issue. Legitimacy of knowledge only arose a few times in the 23 interviews I conducted with land use decision-makers in California, often in the form of local knowledge about ecosystems held by farmers. This less significant role for legitimacy could have been because the Healthy Lands, Healthy Economies Initiative did not focus centrally on incorporating diverse perspectives to produce unbiased assessment outputs, whereas much of the Natural Capital Project’s work emphasizes informing and supporting the processes of land use decision-making across many different decision contexts. Also, the people I interviewed in California were relatively local in terms of the scale of their conservation decisions, so their pre-existing connections with other local groups and sources of knowledge could have made legitimacy less of a prominent issue for them. Or, perhaps the way legitimacy is defined and cultivated is different enough between California and the international Natural Capital Project cases that legitimacy was not always seen as a critical factor.

The salience, credibility, and legitimacy of a body of information are ultimately a matter of perception. They are also claims about and on knowledge, and in this way they are very much a matter of power and politics. Knowledge is contested and constantly negotiated. Knowledge producers, holders, distributors, and users are all enmeshed in political processes of evaluating what is true and deciding on the supposedly best way forward for society. Credibility and legitimacy, especially, are claims on knowledge that

can be used to bolster political positions or to deny the arguments of others. For example, in the case of climate change in the U.S., questioning the credibility and legitimacy of climate change knowledge and climate scientists themselves has been used to justify inaction. Similar claims about the credibility and legitimacy of ecosystem services knowledge are made in: assessments of pollution damage costs (i.e. who pays damage costs for oil spills, how much, and to whom); decisions whether to allow development in ecologically sensitive areas (i.e. should we protect wetlands, or require mitigation for their disruption); and decisions to grant permits to extract natural resources (i.e. where can companies extract natural gas or mine rare earth minerals, and what are the impacts/benefits of these decisions to different groups of people over varying time scales?). Who defines the attributes of knowledge used in these kinds of decisions? In terms of legitimacy, who has say about who gets invited to meetings and which points of view are represented in the process of producing or using knowledge?

These are important questions for ecosystem service proponents to consider. The basic idea of natural capital does not resonate with some populations. Indigenous populations may have established and effective ways to understand and interact with nature, and their livelihoods and cultural traditions can be threatened by the application of ecosystem services to the land. Applications of ecosystem services need to consider how to incorporate traditional knowledge into assessments and how to ensure that a framework designed for conservation is not also systematically providing advantages to one group over another, which can happen when finance mechanisms such as payments for ecosystem services are established. The inclusion of diverse perspectives in ecosystem service projects can take time and effort, but it is essential to consider these

issues of power, privilege, equity, and diversity when generating knowledge that may be used to influence policy.

5.2. Capacity Building

In the course of my research, I considered whether the ecosystem services projects I assessed were really helping those who most need the help. In California, conservation is relatively well established and supported, whereas in Colombia, there may be other more pressing issues that take precedence. The Natural Capital Project attempts to make ecosystem service science available to a broader set of people, but it is still biased in the places it works and the partnerships it develops. It is difficult to accurately assess a place's level of environmental and human development problems, as well as a place's capacity to address environmental and development challenges. But it is clear from the chapter on "global usage of ecosystem service models" that the majority of InVEST model usage is in the U.S., followed by the UK, China, Germany, and France. These countries are not representative of much of the rest of the world, and they are not the countries with the most need for ecosystem service decision support (though it could be argued that China does have high need for this kind of decision support because of population pressures and environmental problems that are significant when compared with other countries around the world). What can be done about this?

The long game is to build capacity for multiple approaches to understanding and addressing environmental challenges. An ecosystem services approach, one of many such options, is being demonstrated and proven in a diversity of contexts including countries with development assistance needs. Building capacity in places with high development

needs could enable a shift in human-nature interactions before significant exploitation of natural resources occurs. Building capacity in places with less development needs could reinforce certain inequities between countries. This issue can be mitigated by selecting locations for trainings that are accessible to diverse participants; offering scholarships to support participation in trainings or ecosystem service applications; and designing strategies to provide equal access. Capacity building can happen through trainings, development and technology assistance, and targeted decision support that engages in deliberative processes and includes diverse stakeholders. The application of ecosystem services has serious limitations, but it also has great potential to inspire collective action for more sustainable forms of development.

5.3. Lessons Learned

In the process of studying how ecosystem services knowledge impacts decisions, I have learned several important lessons. In general, there is a clear need to more carefully evaluate the social-ecological impacts of conservation work. We have a limited understanding of how conservation science becomes used in making policy decisions. When conservation action does occur, we often have incomplete or anecdotal evidence about how projects, programs, or policies perform. We need to feed better evidence of impact into a virtuous cycle in which we test different conservation strategies, systematically learn about what works, and continuously improve the practice of conservation.

I have also learned that it is incredibly difficult to track how decision-makers use particular kinds of knowledge. Many factors beyond scientific knowledge go into land

use decisions – potential for profit, voting and elections, cultural traditions, immediate and pressing priorities such as roads, food, human health, etc. – and it can be difficult if not impossible to isolate a causal role of each of these factors. It makes sense that this area of research has been understudied for ecosystem services.

The impact of knowledge is not just about content of the knowledge. It is about people's relationships with knowledge, and the processes through which their relationships develop. The data for studying these relationships are often poor. Integrating impact evaluation into the design of ecosystem service programs will improve data availability. Baseline data collection is key and sets the stage for subsequent evaluation. In situations with existing long-term datasets, statistical matching techniques between treated and non-treated units can uncover some of the causal pathways through which ecosystem services projects make a difference.

The science and policy of ecosystem services would benefit from consensus on clear, measurable ecological and social outcomes. Consensus on what it means for a program to be successful will enable more consistent, rigorous studies of impact. Clear ideas of success and how to measure it can also be used to test principles and guidelines that are important for the success of ecosystem service interventions. In the last several years, there has been a growing focus on identifying and validating the “enabling conditions” for success, which can only be done with a clearer vision of the outcomes that constitute success. Success might look like stabilized populations of species of concern within some agreed-upon baseline range; decreasing rates of land use change and resource consumption; positive trends in environmental quality over time as measured by levels of key air and water pollutants; or maintenance of human activity within a safe

operating space as defined by Rockstrom's planetary boundaries. Not all forms of success need to involve quantitative metrics, but in order for us to gauge progress toward or away from a goal we must be able to assess progress consistently across projects. The "pathways to impact" diagram is a useful conceptual framework in this regard.

An important aspect of exploring what enables program success is to consider barriers to success and knowledge uptake. Recognizing barriers and admitting where ecosystem service interventions have been less successful (or not worked at all) could provide valuable lessons for future efforts. Practitioners and funders are understandably averse to talking about project failures. But the conservation community can cultivate a culture of learning and develop capacity for future success by documenting these cases and honestly sharing about pitfalls and mistakes. Genuine curiosity about the social-ecological impacts of conservation programs should uncover accounts of failure, not just stories of success.

Incorporating these lessons into conservation work is a challenge. Any organization struggles with how evaluation can divert precious, limited resources away from actually doing projects. And often, it is not easy to evaluate real-world ecosystem service projects across a variety of cultures and decision contexts. Despite the challenges in evaluating the impact of ecosystem services knowledge, it is still worth doing for the conservation community. Conservation organizations need to evaluate their own impact honestly. They could also invest in reliable third party reviews to provide less biased appraisals. In order to do this well, the conservation community would benefit from first defining clearer visions of success (goals), then agreeing on how we would know if we're

making progress toward goals or not (evaluation), and finally deciding what programs and projects to implement (action).

Other fields of study can offer valuable insights. Quasi-experimental methods from policy evaluation are applicable to many efforts aimed at developing ecosystem services policy. The international development world has an established culture of specifying program outcomes and evaluating the performance of individual interventions and larger-scale programs. Education scholars regularly evaluate the social impacts of programs on different populations. Behavioral economics and psychology could illuminate how and why human decisions about land use are made in various contexts. Anthropology and ethnographic studies are vital to uncovering the cultural and social forces at play in a particular place and at the intersection of the science and policy spheres. And complexity research, with its increasingly relevant findings about the dynamics of complex social-ecological systems, could help understand governance networks in PES programs or improve the accuracy of ecosystem service models.

5.4. Concluding Thoughts

At this point in the evolution of the ecosystem services field, there are hundreds of case studies, sound theory about the production, flow, and distribution of ecosystem services across landscapes, an academic journal dedicated to the topic, and increasing demand for ecosystem service assessments from decision-makers. As the field matures, it is important to undergo an intentional process of self-reflection, in the interest of consciously evaluating whether hoped-for impacts are being realized. Taking stock involves answering the fundamental question of how to know whether ecosystem service

projects are successful – both in terms of improving decisions and generating positive environmental and human well-being outcomes. With a clear idea of what success looks like and how to evaluate it, the ecosystem services community can make measureable progress toward conservation goals and move forward with confidence that programs are achieving the intended positive impacts.

As I conclude this dissertation, humans are rapidly changing the planet and scientists are doing an excellent job documenting declines in global environmental quality. But while we increasingly crowd the planet and impact the water, air, and land, we still depend on the diversity of life and nature to support both our immediate well-being and our long-term survival. Now we need to draw sharper connections between declines in environmental quality, proposed causes of environmental decline, and importantly, proven strategies for improving the interactions between people and nature. In order to move toward ecological sustainability, it is critical that we imagine, understand, and expand the beneficial relationships between people and the environment.

The concept of ecosystem services contributes to sustainability by providing a much-needed link between humans and the environment. By explicitly measuring and mapping the flow of benefits from the environment to people, ecosystem service projects can illuminate positive interactions between people and the Earth. While the idea of ecosystem services has an anthropocentric focus on the benefits people receive from nature, it also suggests there are benefits that nature receives from people.

Our choice of perspective dictates our relationship with the environment. A focus solely on the one-way flow of benefits to people is incomplete. Reframing ecosystem services in the context of the whole system underscores people's reciprocal connections

with the natural world. Bringing a deeper sense of interdependence into the ecosystem services field provides an opportunity to enrich these connections, and an invitation for people to live and work in service of ecosystems.

APPENDIX A: FULL LIST OF REFERENCES

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